THE CHEETAHS OF THE NORTHERN TULI GAME RESERVE, BOTSWANA: POPULATION ESTIMATES, MONITORING TECHNIQUES AND HUMAN-PREDATOR CONFLICT

A thesis submitted in fulfilment of the requirements for the degree of

MASTER OF SCIENCE

of

RHODES UNIVERSITY

By

ELEANOR I. BRASSINE

December 2014

Dedicated to the memory of

Joseph Dudley Dexter

First cheetah captured by camera trap in The Northern Tuli Game Reserve, Botswana.

ABSTRACT

Remaining viable cheetah (*Acinonyx jubatus*) populations in Africa are threatened by direct persecution through conflict with farmers and habitat degradation and fragmentation. Botswana is considered a stronghold for free roaming cheetahs in Africa, yet the country has had relatively limited research on its cheetahs, and information from the east of the country is lacking. Data on the current status of populations is thus required to make informed management decisions. My study provides estimates of population density, abundance, distribution and status for the demographically open cheetah population of the Northern Tuli Game Reserve (NOTUGRE) in Botswana. The effectiveness of two population monitoring methods, namely camera trapping and a photographic survey, were also investigated. Moreover, I report on the level of conflict between livestock farmers and predators on rural communal farmlands within and adjacent to NOTUGRE. Data were collected between May 2012 and November 2013. Results indicate a low population density of 0.61 ± 0.18 adult cheetahs per 100 km² and a minimum population size of 10 individuals (nine adults and one cub). Camera traps placed at cheetah scent-marking posts increased detection rates and provided ideal set up locations. This approach, together with Spatial Explicit Capture-Recapture (SECR) models, is recommended for future studies. The long-term studies that are required to better understand the status of cheetahs in Botswana do not exist. Thus, photographic surveys may provide an alternative method for providing baseline data on population numbers, distribution and demography. The third aspect of my study gathered information on levels of livestock loss and human tolerance of predators through the use of interviews (n = 80). Conflict with subsistence farmers is a concern as livestock depredation is relatively high (9.1% of total livestock owned) and farmers had an overall negative attitude towards conservation of large predators. My results suggest that human-predator conflict in this area is more complex than the direct financial loss from depredation. Hence, reducing depredation rates alone is unlikely to change farmer tolerance of wildlife on farmlands. Improved, responsible farm management, including self-responsibility for livestock rearing, and positive appreciation for wildlife are necessary. The NOTUGRE cheetah population requires further research to understand possible threats to the population. Furthermore, a better understanding of the connectivity between cheetahs of NOTUGRE, South Africa and Zimbabwe is required. The number of cheetahs within NOTUGRE is too small to sustain a viable population, hence conserving cheetahs outside of the protected area should be a priority for the conservation of the

population. This can only be achieved through assistance and involvement from national authorities, local people and conservation organisations.

ACKNOWLEDGEMENTS

As I embarked upon this project, it became readily apparent that its success would rely heavily on the support of many people and organisations. I am truly grateful to all of you that believed in me and decided my research was worth your support. In particular, I would like to acknowledge and thank the following:

The National Geographic's Big Cat Initiative for funding the project, I would like to particularly thank Dr Luke Dollar for his continued assistance with this grant.

The Rufford Foundation's small grant program for funding the project.

Zoofaris and Wildwonders, but more specifically, Jackie Navarro, director of Wildwonders, for not only providing funds for this project but also promoting my project overseas to raise the funds. Thank you for your support!

The financial assistance of the National Research Foundation (NRF) towards this research is hereby acknowledged. Opinions expressed and conclusions arrived at, are those of the author and are not necessarily to be attributed to the NRF.

Rhodes University Postgraduate Scholarship, with this I would like to particularly thank John Gillam and his team for their assistance in securing my two bursaries.

Mashatu Game Reserve for providing accommodation, a research vehicle and administrative help. I am particularly grateful for the use of a Land Cruiser, the best vehicle a researcher could ask for. It never failed to surprise me where this vehicle could take me! It was a great privilege to work and live at Mashatu. In particular, I would like to thank David Evans and Pete le Roux, for presenting me with the opportunity to conduct research in The Northern Tuli Game Reserve and granting me permission to reside and operate at Mashatu. Also Pete, for your help in starting up the Northern Tuli Cheetah Project, and for always finding the time to answer to my emails.
I will always be thankful to the NOTUGRE land owners for granting me permission to conduct research in the reserve and for the donation of cool drinks for participants during my questionnaire survey.

Tuli Safari Lodge for generously sponsoring two cheetah satellite collars, I would like to particularly thank Will Roughton and Nick Hilterman for the interest you showed in my work and your continuous help in finding funds for the project.

Idea Wild for sponsoring camera traps and batteries.

The Wildlife and Reserve Management Research Group for supplying equipment including camera traps and casings.

The Department of Wildlife and National Parks of Botswana and the Government of Botswana for granting me permission to conduct research in Botswana.

Dr Dan Parker, my supervisor, not only for your academic guidance, but for your continued encouragement and intellectual and moral guidance when I needed it most. Thank you for your friendship and giving me unlimited intellectual freedom and for the support you showed when I decided to take on this research. I would also like to thank you for getting my drafts back to me in record speed!

Everyone who helped me in the field. I am most particularly grateful for your company as long hours in the field alone can be monotonous and, at times, tedious. I have also learned that an incredible wildlife sighting is not the same when you do not have someone to share it with. Mike

Dexter, you were by far my most valuable assistant and this project could not have been realised without your continuous assistance and support. I would also like to thank the following people who helped me with field work: Gwendoline Brassine for long hours in the Botswanan sun whilst setting up camera traps; Tiro Maleke, Lauren Satterfield, Kaizer Sekanonyana, and Tim Forssman for your assistance with my questionnaire survey.

Larry Patterson and Erik Verryen for providing veterinary work and for your expertise.

The photographic survey could not have happened without the contribution of photographs from a number of people. Above all, I would like to thank all the professional photographers, not only for contributing beautiful sharp images but also sharing with me your unedited images from your archives.

The Mashatu Guides, in particular Fish Maila, Kaizer Sekanonyana, Bashi Patane, Elvis, Dan Masupe and Richard Modeme for volunteering to help me with my research by either pointing out cheetah scent marking posts or taking photographs of cheetahs whilst on game drives
Louise & Core Carelsen, Kate Barrow, Saskia von Sperber for providing me with company and moral support when times were difficult.

Marco & Vivian Harms and Ian & Charlotte Pollard for your friendship and support, providing me with unfailing hospitality

Adolf for servicing and repairing my research vehicle. You were always ready to help me out at the drop of a hat!

Shem Compion and Zendré Lategan for your hospitality and for providing us lay over accommodation and great company on numerous occasions.

Tico McNutt from The Botswana Predator Conservation Trust, for your assistance with some of the survey designs and help in securing funding for the project.

A number of people provided valuable information on cheetah sightings; specifically I would like to thank Stuart Quinn, Bruce Petty, Cliffy Phillips, Brian Rode and Chantelle Venter. Although you all lived far away it was always a pleasure to see you after a long dusty (or muddy) drive.

Mike Lester, David Wyman, and Doug Devries for your help to locate Mapula, the radiocollared cheetah that had been missing for a few months in an unnerving yet spectacular flight over the reserve.

Martin Villet for reading through the drafts of this project and providing moral support.
Mathilde Brassine for your assistance with bursary applications and reading the draft of this thesis.

Armand Kok and Jeff Jenness for your assistance with GIS whilst I was in Botswana.

Andy Royle for your assistance with SPACECAP.

Travis Perry for your help with statistical analyses

Galagos Wildlife-Conservation for the Elephant proof cement blocks, an ingenious idea to protect the camera traps.

I am also grateful to Brenda & Peter Dexter for your support throughout the years, you have always been there when we needed someone to fall back on. Thank you.

My love of the bush and wildlife could not have developed without my incredible upbringing from my parents, Manu Brassine & Nicole Dehalleux. From an early age my sisters and I were taken to some of the most remote places in Africa, experiencing the true African bush as we would set off for extended holidays, I could not have asked for a better childhood! Through your love and support I have been able to pursue my dreams.

Most of all I would like to thank Mike Dexter, for your continuous and persistent motivation throughout this journey; you were my assistant in the field and later in the writing up. You relocated to Grahamstown with me and provided the continuous support when I needed it most and always acknowledged my success. Thank you for sharing with me some unforgettable experiences in the bush.

Thank you to all my friends and work colleagues who helped proof reading my thesis and discussing some of the findings of this project and to anyone I may have accidentally left out of this list.

TABLE OF CONTENTS

Chapter 4. Citizen Science in Cheetah Research

Chapter 5. Human-Predator Conflict

CHAPTER 1

GENERAL INTRODUCTION

1.1 Cheetah global status

The cheetah (*Acinonyx jubatus*) is currently listed as vulnerable by the International Union for the Conservation of Nature (IUCN), Global Red List (IUCN 2013) and figures as an Appendix I species on the Convention on International Trade in Endangered Species of Fauna and Flora (CITES). In addition, available data suggests that cheetahs are close to being classified as endangered and Ray *et al.* (2005) identify the cheetah as a species in crisis with a high overall level of priority based on its high vulnerability and exposure to a suite of external threats. The cheetah used to be widely distributed across Africa and south west Asia, however in the past few decades the species distribution range and total numbers have reduced dramatically from approximately 15 000 in the early 1970s (Myers 1975) to a maximum of 12 000 by 1998 (Marker 1998) and more recently (2008) to approximately 7500 individuals (Buk & Marnewick 2010). However, from a species conservation perspective, it is important to note that it is the remaining viable sub-populations that should be considered rather than just individual animals (Marker 1998). Viable cheetah populations are only found in sub-Saharan Africa with Tanzania, Kenya, Namibia and Botswana considered strongholds (Marker 1998).

1.2 Threats to cheetah populations

The critical status of the cheetah is based on two main components. Firstly, the cheetah is highly vulnerable and this is mostly due to extensive range loss (>75% of their range in the last 150 years), a relatively high degree of specialisation and low reproductive rates (Caro 1994; Ray *et al.* 2005). Secondly, cheetahs are exposed to a high number of threats which are contributing to the species' global decline (Ray *et al.* 2005). The main threats identified are habitat loss, conflict with livestock farmers and competition with other large predators (Myers 1975; Ray *et al.* 2005).

Human persecution

The greatest threat to cheetahs is arguably their direct persecution by humans either from exporting of live animals or killing (Woodroffe & Ginsberg 1998; Ray *et al.* 2005; Marnewick *et al*. 2007; Purchase *et al*. 2007; Marker *et al.* 2010). Cheetahs are often eliminated on livestock farms as they are perceived to pose a threat to livestock (Marker *et al.* 2003a; Ray *et al.* 2005;

Holmern *et al.* 2007). Game farmers, who rely on live sales of antelope or trophy hunting, will also often consider the cheetah a liability (Marker *et al.* 2003a; Marnewick *et al*. 2007). Cheetahs are eliminated mostly by shooting on sight, but also by vehicle collisions, trapping with cages and then shooting trapped animals, snaring and poisoning (Ray *et al.* 2005; Marnewick *et al*. 2007; Marker *et al.* 2010). The wild cheetah population also suffers from over-exploitation, with cheetahs being captured and exported both legally and illegally to international captive facilities (Klein 2007; Marnewick *et al*. 2007; Purchase *et al*. 2007).

Habitat loss

Cheetah populations suffer from the loss of natural habitat and habitat fragmentation resulting from human encroachment and development linked to an increasing human population (Meyers 1975; Woodroffe & Ginsberg 1999; Marnewick *et al*. 2007; Macdonald *et al.* 2010a). Furthermore, an increasing human population is often accompanied by the depletion of the prey base through over-exploitation or habitat destruction (Kelly & Durant 2000; Broomhall *et al.* 2003; Ray *et al.* 2005; Marker *et al.* 2010). Human encroachment can also alter the habitat in terms of bush encroachment and the introduction of artificial waterholes which can cause a change in prey species composition and may make the habitat less suitable for cheetahs (Buk & Marnewick 2010). For example, accessibility to water causes an increase in antelope densities and consequently higher densities of larger predators such as lions (*Panthera leo*) and spotted hyenas (*Crocuta crocuta*) (Buk & Marnewick 2010). Cheetahs prefer areas of relatively low prey density that are normally avoided by other large predators (Durant 1998), a predator avoidance strategy to reduce kleptoparasitism and cub mortality (Durant 1998), so these changes in habitat may be detrimental.

Competition with large predators

The viability of a cheetah population is also affected by the density of other large predators (Caro 1994). Cheetahs are subjected to a high rate of intra-guild competition and kleptoparasitism from larger carnivores such as lions and spotted hyenas (Marnewick *et al*. 2007; Houser *et al.* 2009; Durant *et al.* 2010). Larger predators also contribute to cheetah cub mortality and in some instances adult mortality (Caro 1994; Laurenson 1994; Laurenson *et al*. 1995; Kelly & Durant 2000; Ray *et al.* 2005; Macdonald *et al.* 2010a), hence cheetahs are often more successful outside protected areas where other large predators have been extirpated or occur at lower population densities (Laurenson *et al*. 1995; Marker 1998; Marnewick & Cilliers 2006). Outside of reserves, however, cheetahs frequently come into contact with livestock farmers, who may consider them a threat to their livelihoods leading to their persecution as mentioned previously (Marker 1998; Woodroffe & Ginsberg 1998; Selebatso *et al.* 2008). The amount of potential habitat for cheetahs, including protected and un-protected areas is, therefore, influenced by, but not limited to, the population status of other large predators and the level of human activity.

1.3 Conservation of cheetahs in Botswana

Although there have been detailed cheetah population studies conducted in East Africa (Caro 1994; Gros 1996, 2000, 2002; Kelly *et al*. 1998; Kelly & Durant 2000) and Namibia (Marker *et al.* 2003b, 2007, 2008a, 2008b), significant knowledge gaps still remain for many parts of their range (Ray *et al.* 2005; DWNP 2009). Botswana has had relatively limited research on its cheetah populations, and information specific to the east of the country is sorely lacking. However, Botswana is believed to be a key country for the remaining viable populations of cheetahs, holding the second largest population of cheetahs in southern Africa after Namibia (Marker 1998; Purchase *et al.* 2007; Winterbach *et al.* 2014), with a national population estimate of 1768 cheetahs (Klein 2007). Furthermore, the Botswana cheetah population is believed to be a large contiguous population with the cheetah populations of South Africa, Namibia, Zambia and Zimbabwe (DWNP 2009).

Botswana has designated approximately 17% of its land to wildlife protection (Game Reserves and National Parks) and an additional 21% designated as Wildlife Management Areas (WMA) where sustainable wildlife use is permitted (Klein 2007). However, cheetahs are found throughout the country, with about half of the cheetah population occurring outside of formally protected areas (Myers 1975; Marker 1998; Winterbach *et al.* 2014). Therefore, agricultural zones are important areas for cheetahs and conservation efforts should encourage the co-existence of cheetahs and humans (Caro 1994; Winterbach *et al.* 2014). Livestock farming in Botswana has grown exponentially in the last few decades along with the accompanied change in land use from previously unoccupied wildlife areas to livestock farmlands (Klein 2007). The national livestock herd is estimated at 4.7 million (DWNP 2012) yet the maximum sustainable herd was evaluated at 3.3 million cattle (*Bus taurus*) (World Bank 1983). Lack of livestock management in Botswana has resulted in deterioration of the veld (open landscape covered in grass or low scrub) and habitat degradation is evident by the decline in perennial grasses, lowered water tables, widespread thorn bush (*Acacia* spp.) encroachment and an overall decrease of wildlife (Myers 1975; Klein 2007). Veterinary cordon fences, to control the spread of foot and mouth disease, have further aggravated the situation as these barriers can prevent the natural movement of wildlife (Bartlam-Brooks *et al.* 2011; Cozzi *et al.* 2013). Additionally, the increase in the human population and pastoral activities in previously uninhabited wildlife areas has led to an increase in human-predator conflict particularly as a result of an increase in encounter rates between livestock and predators (Klein 2007). Human-predator conflicts can have severe negative consequences to large predator populations due to direct persecution by farmers (Ogada *et al.* 2003; Thorn *et al.* 2014).

The Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES) has granted Botswana five cheetahs for annual export as live specimens and hunting trophies. Despite this cheetahs are legally protected under Botswanan legislation and may not be killed under any circumstances, however the species suffers from both illegal poaching and persecution by livestock farmers (Klein 2007). The protection of remaining viable populations of cheetahs requires their conservation outside protected areas, particularly populations which straddle international boundaries and experience different acting laws and persecutions (DWNP 2009).

1.4 Study rationale

Despite the importance to conservation and management planning, the status of the cheetah in Botswana is poorly researched. Klein (2007) provides a summary of the past cheetah research undertaken in Botswana: Population censuses have been carried out in the Central Kalahari Game Reserve (CKGR), the Kgalagadi Transfrontier Park (KGTF), Ramsar Site in the Okavango Delta, and Jwana Game Park through the use of spoor surveys (Klein 2007). More recently, follow up studies have been conducted in CKGR (2012), KGTP (2013) and areas around the CKGR (2014) (R. Klein, Cheetah Conservation Botswana, pers. comm.). Moreover, a camera trapping study is currently being carried out in the Ghanzi farmlands area (R. Klein pers. comm.). Information on the status and distribution largely comes from interviews, opportunistic sightings, and Problem Animal Control (PAC) reports. Of particular concern is that the focus of cheetah research has only been carried out in certain areas of the North, Central and South of the country (Figure 1.1 taken from Klein 2007) and information on the status of cheetahs, including estimates of total cheetah numbers in Botswana, is derived from two spoor surveys undertaken in the CKGR and KGTF (Klein 2007) (Figure 1.1). Furthermore, monitoring programs to determine population trends have yet to be conducted, with the only information on population trends obtained from a status questionnaire survey conducted by Cheetah Conservation Botswana (CCB) in 2006 (Klein 2007). Although information from such a survey can provide quick and useful baseline information, it does not replace the need to establish adequate monitoring programs to understand trends in cheetah populations and possible threats to these populations (Gros *et al*. 1996).

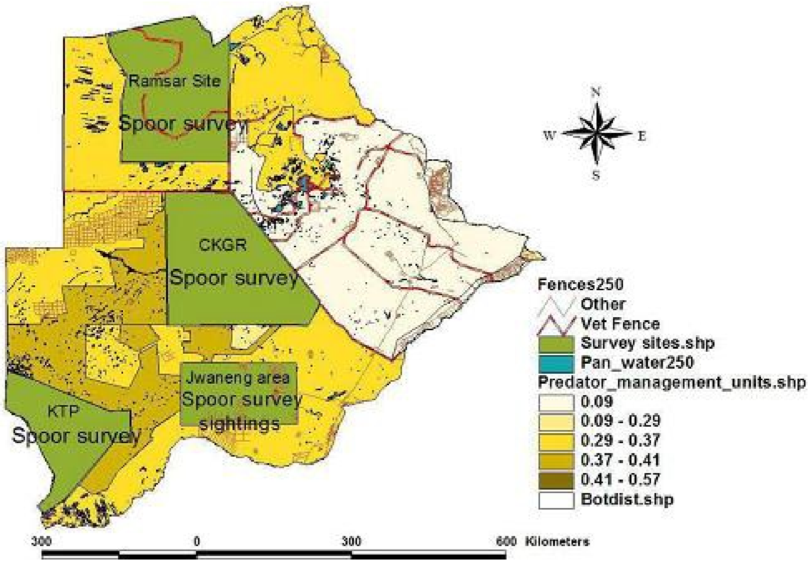


Figure 1.1 Map of Botswana showing regions (in green) which have had more focused cheetah research and derived density estimates (Reproduced from Klein 2007).

Population size and trends in population sizes are recognized as the most important predictors of species extinction risk (O'Grady *et al.* 2004), yet there is clearly a gap in our knowledge of the population size and status of Botswana's cheetahs, particularly in the east of the country where research has mostly been absent (Figure 1.1). Thus, the cheetah population of Botswana requires more in-depth information on population sizes and distribution in different habitat types and land use areas (Klein 2007; DWNP 2009). In addition, assessments of the impact of predator-conflict on cheetah populations in communal farmlands are urgently needed (Klein 2007).

In this study, I provide information on the status of cheetahs in the most eastern region of Botswana. Information on the occurrence, distribution, population size and density estimates, and apparent trends in numbers are documented. I also report on population demographics and, where feasible, estimated age and family relations of specific individuals. My study also seeks to develop an effective monitoring tool for cheetahs by addressing the efficiency of various field methods and sampling designs to effectively monitor cheetah populations. Specifically, I evaluate the suitability and effectiveness of camera trapping surveys and photographic surveys for providing quick and reliable estimates on cheetah population status, size and density. Finally, my study documents human-predator conflict within the livestock farming communities bordering the Northern Tuli Game Reserve (NOTUGRE) in Botswana.

CHAPTER 2

STUDY AREA

2.1 Location

The study was undertaken in the Northern Tuli Game Reserve (from here on referred to as NOTUGRE), a private game reserve situated in the eastern corner of Botswana. The region lies between latitudes 21º55' and 22º15'S, and longitudes 28º 55'and 29º15'E (Figure 2.1) and forms the eastern limit of The Tuli Block, a 350 km strip of privately owned land located north of the Limpopo River (McKenzie 1990).

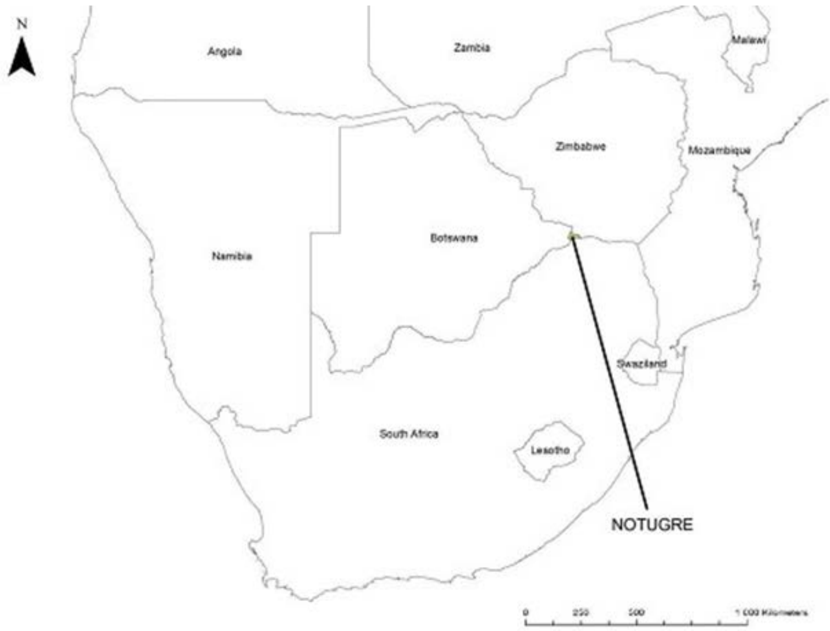


Figure 2.1 The location of the Northern Tuli Game Reserve (NOTUGRE) in eastern Botswana. (ArcGIS 10; map units: decimal degree; not projected).

NOTUGRE is naturally delineated by the Shashe River in the east and the Limpopo River in the south (Figure 2.2). The former forms the border between Botswana and Zimbabwe and the latter, the border between Botswana and South Africa. The South African border is fenced (total fence length ~86km) but poorly maintained and does not restrict animal movement (Jackson *et al.* 2012). The northern boundary consists of a cut-line demarcating the Tuli Circle Safaris Area in Zimbabwe. Animals move freely across this boundary and there is limited human activity as the area is only used seasonally for trophy hunting purposes. On the western boundary there is an electrified game fence (height: 2.1m; 3 electrical stands at 1.8m, 50cm, 20cm) intended to prevent wildlife movement out of the reserve as well as livestock into the reserve. However, it is frequently damaged by elephants (*Loxodonta africana*) and other wildlife and therefore does not normally restrict the movements of large carnivores and/or livestock. The south-western and eastern boundaries are unfenced. The study area also has a double veterinary cordon fence which runs north to south in the west of the reserve (Figure 2.2). This fence was built to control foot and mouth disease by preventing large herbivore movements (Kgathi *et al.* 2012). However, small ungulates and some large ungulates, such as kudu (*Tragelaphus strepsiceros*) and eland (*Tragelaphus oryx*) are able to cross this fence (pers. obs.). The game fences of NOTUGRE also do not restrict movement of large carnivores; cheetahs (*Acinonyx jubatus*), lions (*Panthera leo*), leopards (*Panthera pardus*), spotted hyenas (*Crocuta crocuta*) and African wild dogs (*Lycaon pictus*) frequently move across these fences (Jackson *et al*. 2012).

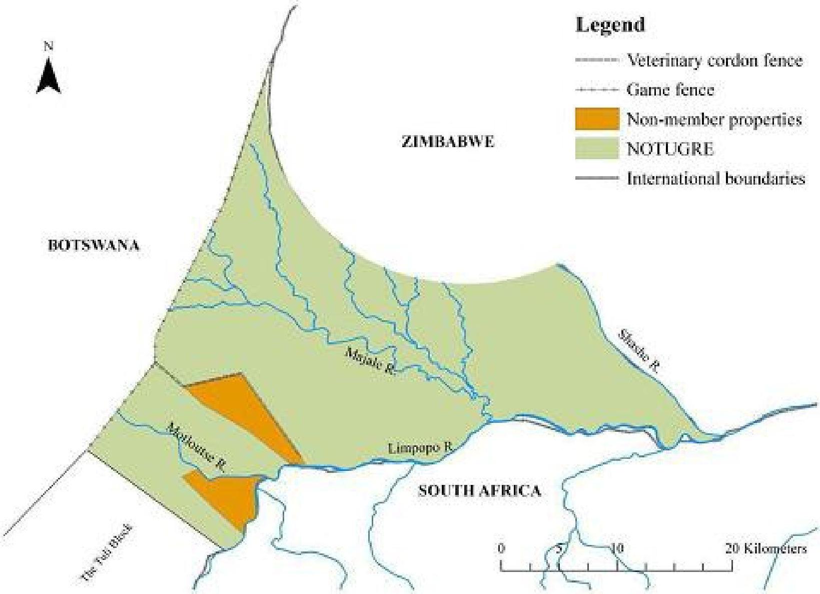


Figure 2.2 Map of NOTUGRE (green) with the major drainage channels. A game fence runs along the western boundary (single dashed line) and a poorly maintained fence runs along the southern banks of the Limpopo River. There are no fences along the Zimbabwe boundaries. A double veterinary cordon fence (double dashed line) runs north to south in the west of NOTUGRE. Non-member properties (Portion 2 of Lowensa-la-Moridi and Talana Farm) are shown in orange. (ArcGIS 10; map units: decimal degree; not projected).

2.2 Land use

NOTUGRE was established by multiple landowners in 1986 (Steyn 2004). It consists of 36 individual properties and encompasses an area of 728km². These individual properties are used for commercial ecotourism or private holiday purposes (Steyn 2004). The aim of the reserve is the conservation of wildlife. There is no farming or pastoral activity in the reserve (Steyn 2004), although one of the properties has some livestock and a citrus orchard (Fairfield property) which are enclosed within a game fence. Within the NOTUGRE boundary, there are two non-member properties (Figure 2.2); Portion 2 of Lowensa-la-Moridi and Talana Farm. The former is situated north of the Limpopo River and west of the veterinary cordon fence. A small village, Lentswe Le Moriti, is located in the south eastern corner of the property and the land is also used for crop and pastoral farming. Talana farm is an agricultural farm situated west of the Motloutse River and along the banks of the Limpopo River. This property is surrounded with an electrified game fence.

Adjacent communal farmlands north west of NOTUGRE also formed part of the study area. The main form of land use on the communal farms is subsistence agricultural and livestock pastoralism; livestock kept includes goats (*Capra hircus*), sheep (*Ovis aries*), cattle (*Bus taurus*) donkeys (*Equus asinus*) and chickens (*Gallus gallus domesticus*). Subsistence agricultural and livestock farming also occurs east of the reserve across the Shashe River in Zimbabwe. South of the Limpopo River are privately owned South African farms which are used for commercial crop farming, sport hunting, and game and livestock farming. All farms in South Africa are relatively well fenced. Mapungubwe National Park in South Africa also borders onto the Limpopo River to the south of NOTUGRE.

2.3 History

The area has been occupied by human settlements since before 800AD, first with small groups of Stone Age people and then later by Iron Age people (McKenzie 1990). There is also evidence of settlements of the Babirwa Bantu people, who farmed and kept large numbers of livestock until 1926 (McKenzie 1990). Archaeological artefacts found on the reserve suggest intensive pastoralism from between 900AD and the 1800s (Dr T. Forssman, archaeologist, pers. comm.). In the late 1880s and early 1890s, there was large scale movement of European settlers to the area with periodic attempts at cattle farming (Lind 1974). With this influx of settlers came the construction of artificial waterholes, roads, human habitations and an overall increase in human activity (Lind 1974). Large predators (species unspecified) were heavily persecuted during this time, with the highest persecution occurring in areas with higher human activities (Lind 1974). In the 1950s, at least a 150 lions were shot (Lind 1974). In the mid-1960s, farming and most hunting ceased when a number of landowners campaigned for the development of a wildlife sanctuary (Lind 1974). By this stage, most of the fauna of the region had been decimated and large predators were almost extinct (Lind 1974). In 1975, the Northern Tuli Conservation Association was formed with the purpose of conserving wildlife and all hunting ceased by 1987 (McKenzie 1990).

The mammalian fauna have largely recovered but some species did not return to the area, including the white rhinoceros (*Ceratotherium simum*), African wild dog, roan antelope (*Hippotragus equinus*), sable antelope (*Hippotragus niger*) and giraffe (*Giraffa camelopardalis*).

Giraffes and African wild dogs were reintroduced in NOTUGRE in 1984 and 2008, respectively. However, the wild dog population suffered severehuman persecution and as of 2012, no resident wild dog pack remained within NOTUGRE.

2.4 Climate

The climate of NOTUGRE can be described as semi-arid and sub-tropical with temperature fluctuating between -5ºC and 42ºC (McKenzie 1990). Temperatures peak during December and January, and reach their minimum during June, July and August (Figure 2.3). Occasional light frosts have been recorded during harsh winters (Lind 1974). Rainfall is low and unpredictable and the majority falls in the summer months between November and March, usually induced by convectional movements (Figure 2.4). The average annual precipitation is 386.5mm (for the years 1996-2013) with peak rainfall years receiving as much as 917mm (2000) and low rainfall/drought years as little as 172mm (2012) (Figure 2.5). Light showers may occur occasionally in April and September. Rainfall occurs mostly as afternoon thunderstorms with localised showers. However, widespread soaking rains may occasionally occur, but these are uncommon and normally only occur in wetter years (McKenzie 1990). The prevailing wind is south-easterly, with whirlwinds common in the dry winter months (Lind 1974). Data for the below figures (Figure 2.3 – 2.5) were taken from four weather stations within NOTUGRE. Weather stations with incomplete records were removed from the dataset and only used where appropriate.

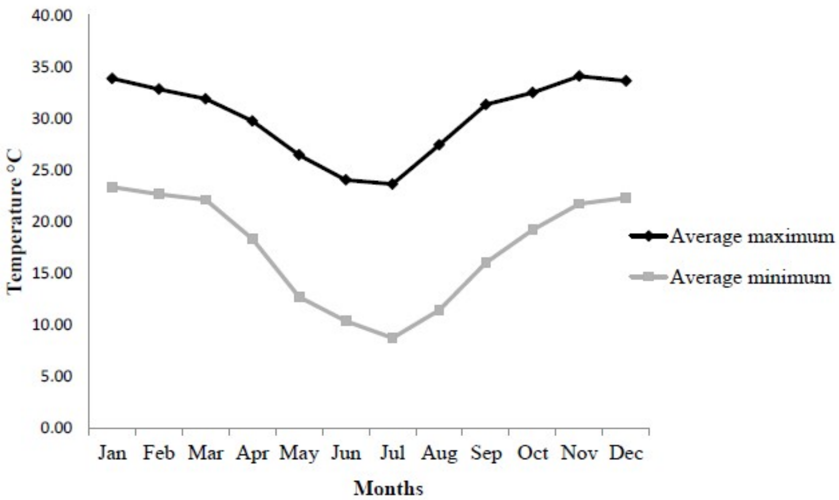


Figure 2.3 The mean monthly maximum and minimum temperatures (°C) for NOTUGRE over a 17 year period (1996-2013) taken from two weather stations (Mashatu Main Camp; Mashatu Tent Camp) within NOTUGRE.

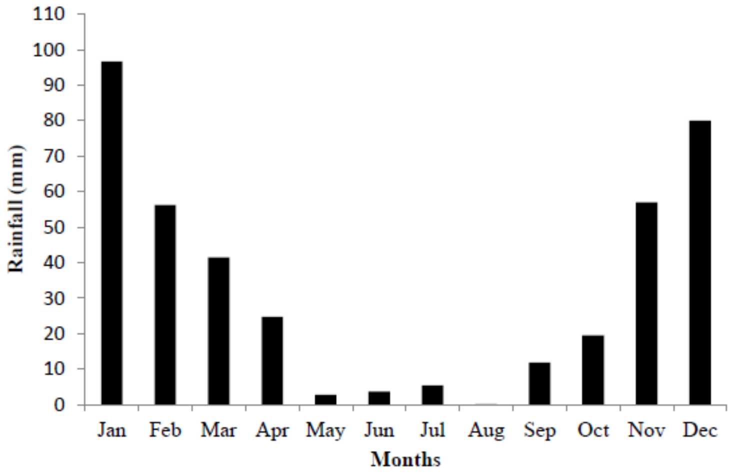


Figure 2.4 Average monthly rainfall for NOTUGRE taken from three weather stations (Mashatu Main Camp; Mashatu Tent Camp; Jwala Lodge) over a 17 year period (1996 – 2013).

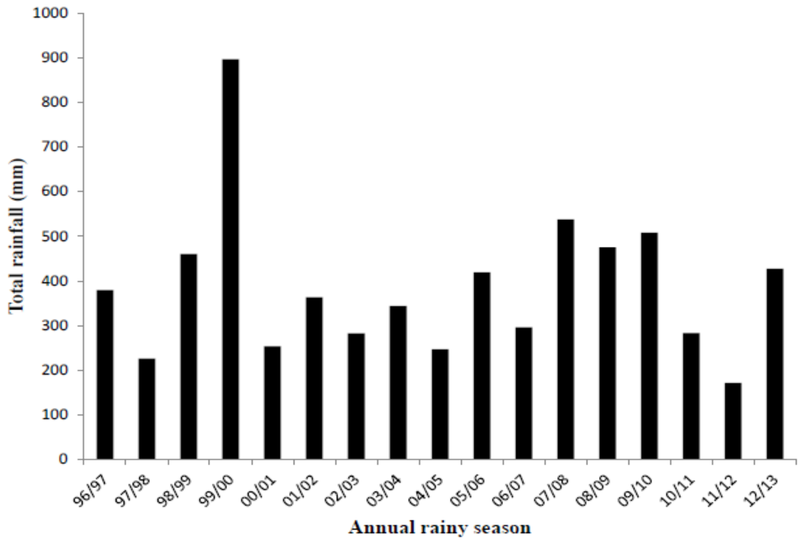


Figure 2.5 Total annual rainfall (mm) for NOTUGRE averaged from four weather stations (Mashatu Main Camp; Mashatu Tent Camp; Jwala Lodge; Limpopo Valley Airfield) from 1996 to 2013. Annual rainfall is calculated over each rainy season rather than the calendar year.

2.5 Topography and Drainage

The study area is bisected by a number of river channels with the run-off of the entire area draining into the Limpopo River (McKenzie 1990) (Figure 2.6). The Shashe and Motloutse rivers are two of the Limpopo River's largest tributaries within the study area (Figure 2.2). The Majale and Pitsane rivers are also major rivers that drain south-easterly into the Limpopo River. Other minor rivers flow directly into the Majale, Motloutse, Limpopo and Shashe rivers. These rivers run in a general north–south direction. All are non-perennial, flowing only sporadically for few hours or days following rainfall during the summer months (McKenzie 1990). During winter, rivers are dry with the exception of isolated pools in some of the major rivers (McKenzie 1990). Artificial waterholes and natural waterholes (n = 56) are also scattered throughout the reserve, although many pump fed waterholes were discontinued following the formation of NOTUGRE (P. Le Roux, Mashatu General Manager, pers. comm.).

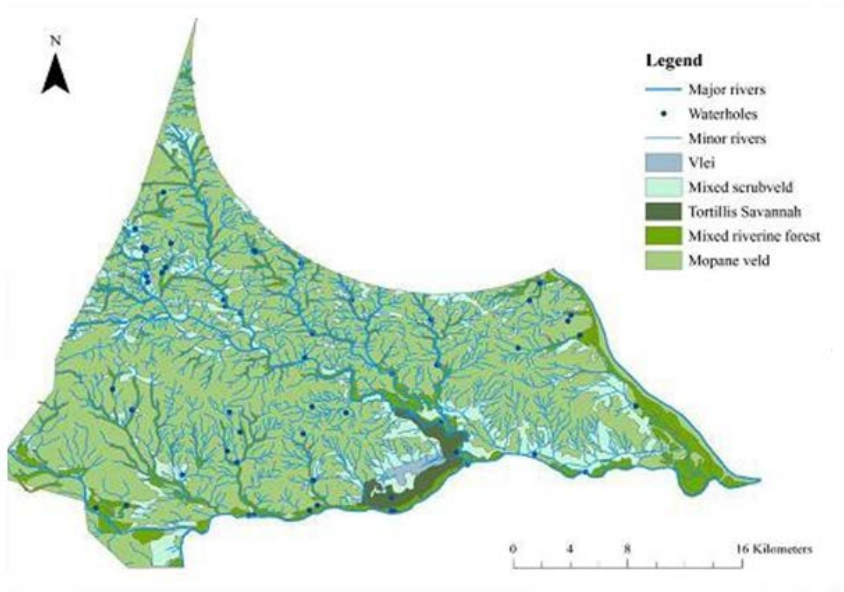


Figure 2.6 Map of the Northern Tuli Game Reserve showing the river channels, main broad vegetation types and waterholes on the reserve (ArcGIS 10; map units: decimal degree; not projected)

NOTUGRE has an average elevation of about 600m.a.s.l. The topography is predominantly flat, particularly around the major rivers. This flat landscape slows the drainage and causes silt deposition and the formation of marshes, locally known as vleis. Two prominent vleis hold water during the summer months; one is situated near the Majale-Limpopo River confluence (Figure 2.7) and the other is found east of the Motloutse River.

Figure 2.7 The vlei near the Majale-Limpopo River. Elephant grass, *Sporobolus consimilis*, forms the main vegetation type. Photo: Mike Dexter

2.6 Geology and Soils

The geological formations of the study area belong to the Basement Complex and Karoo Supergroup, with the basement exposures belonging to the Central Zone or Messina Group of the Limpopo Mobile Belt, with several metamorphosed sedimentary rock types (Joubert 1984). The geology comprises deep Clarens sandstone formations overlain by Letaba and Sabi River basalt formations, cut by a number of east-west dolorite dykes (Joubert 1984). The landscape is relatively flat apart for protrusions of the more resistant dykes that form long narrow ridges intersected by the main river channels (Joubert 1984). Sandstone outcrops, locally known as koppies, also occur along the Limpopo and Motloutse rivers (Figure 2.8) (Joubert 1984). Along all the major river systems are alluvial floodplains which typically have nutrient rich soils (Joubert 1984).

Figure 2.8 A view near the Limpopo-Motloutse confluence showing a sandveld valley and a sandstone koppie with *Acacia* species as the dominant species (foreground). In the background spreads a wide expanse of Mopane Veld with *Colophospermum mopane* as the main woody vegetation. Photo: Eleanor Brassine

Over-grazing of the herbaceous layer by both wild herbivores and livestock has resulted in accelerated and extensive erosion which is particularly evident in the form of sheet and donga erosion and large bare areas completely devoid of vegetation (Figure 2.9) (McKenzie 1990). Basalt areas have a very thin layer of top soils remaining, with Glenrosa, Mispah and Mayo forming the dominant forms, while riverine soils are deep and belong mainly to the Oakleaf, Valsriver and Rensburg forms (Joubert 1984; Mckenzie 1990). Alexander (1984) classified the soil into nine descriptive classes which were subsequently regrouped into three major soil types. These are residual soils, alluvial soils, and eluvial soils.

Figure 2.9 Example of a large bare area completely devoid of vegetation, apart from small clumps of the common annual forb, Dubbeltjie (*Tribulus terrestris*), as seen in the foreground. Photo: Mike Dexter

2.7 Vegetation

NOTUGRE falls within the Mopane Bioregion of the Savannah biome, classified as arid, base rich savannah (Gotze *et al.* 2008). The vegetation can be broadly classified as Mopane Veld, but is also made up of a wide variety of other smaller habitats (McKenzie 1990). The main rivers are flanked by riverine forests forming a thick canopy (Figure 2.10). Species in this habitat include Mashatu trees (*Xanthoceris zambeziaca*), groves of Fever trees (*Acacia xanthophloea*), and Mlala palms (*Hyphaene banguelensis*) (Seliers 2007). Croton (*Croton megalobotrys*) thickets can also be found along the banks of the more prominent water channels including the Majale and Pitsane rivers (Figure 2.10; pers. obs.).

Figure 2.10 Alluvial plains with *Boscia foetida* savannah (foreground) and *Croton megalabotrys* thicket (background) along the banks of the Majale River. Photo: Eleanor Brassine

Fifteen different vegetation types were described by Alexander (1984) based on the species present and their relative abundance, but it is likely that changes in the vegetation have occurred since that study which was of a preliminary nature. Furthermore, distinguishing between the different habitat types can be difficult as they often merge into one another. McKenzie (1990) later regrouped Alexander's 15 vegetation types into either predominantly open or closed. Predominantly open habitats: *Boscia foetida* savannah, *Colophospermum mopane Terminalia prunoides* middleslopes, *Colophospermum mopane* scrubveld, *Salvadora angustifolia* bushveld. Predominantly closed habitats: Valley bush, *Acacia tortillis* savannah, and *Croton megalabotrys* thicket. Joubert (1984) also groups these habitats into three landscapes, namely: Floodplain on alluvium; *Colophospermum mopane*/*Terminalia prunioides* Rugged Veld on Basalt; and Karoo Sandstone landscape. The Rugged Veld on Basalt is the most dominant landscape and occurs in various forms on the reserve. *Colophospermum mopane* and *Combretum apiculatum* form the dominant woody vegetation which falls within the Mopane-Combretum shrub-savanna plant community (Joubert 1984; Nchunga 1978). The Floodplain landscape occurs along the Limpopo, Shashe and Motloutse rivers and is made up of mostly alluvial soils with *Acacia tortillis* savannah (McKenzie 1990). *Acacia albida* Gallery forest and *Croton megalobotrys* thickets also occur in the riverine areas fringed by S*alvadora angustifolia/Acacia tortilis* Bushveld (Joubert 1984). *Boscia foetida* savannah is found on the old undulating floodplains (McKenzie 1990). The Sandstone landscape is found only in the south and west of the reserve, it is composed of sandstone outcrops and sandveld valley composed of sparse woody vegetation and grasses (McKenzie 1990). The vleis are dominated by stands of tall Elephant grass, *Sporobolus consimilis*, which can reach a height of 1.5 to 2m, woody vegetation is absent in this habitat (Joubert 1984). Figure 2.6 shows the main vegetation types found on NOTUGRE together with the water channels and waterholes.

2.8 Fauna

NOTUGRE supports large populations of ungulates particularly in the dry season due to the presence of both natural and artificial water points. Common large and medium mammal species are listed in Appendix I. The reserve has resident populations of all naturally occurring large carnivores, with the exception of the African wild dog which only occurs sporadically. Wildlife is free to move out of the reserve into neighbouring properties where permitting.

The following species used to occur in NOTUGRE but have been absent for over 40 years (Lind 1974):

Alcelaphus buselaphus Red hartebeest
Damaliscus lunatus Tsessebe
Hippotragus equinus Roan antelope
Hippotragus niger Sable antelope
Oryx gazella Gemsbok
Syncerus caffer African Buffalo
Redunca arundium Common reedbuck

A possible reason for the local extinction of these species is the change in vegetation particularly along the river banks which are believed to have been more densely vegetated (Lind 1974). Changes to the environment such as artificial waterholes and human activity brought about by settlers may also have influenced the movement, concentration and general wildlife populations (Lind 1974).

2.9 Study animal

Accounts of cheetahs in NOTUGRE are scarce. Between 1966 and 1971 only two sightings of two cheetahs were recorded (Walker 1971) and there were reports of a number (number unspecified) of cheetahs poached and found at a trading store south of the Motloutse River (Walker 1971). Lind (1974) describes the cheetah as rare and only seen sporadically in NOTUGRE. He gives a few accounts of cheetah sightings and broadly estimates the population to have a total of seven individuals of unspecified ages in 1972 and nine individuals by the end of 1973. Numbers were estimated based on total number of sightings and not individual recognition. Lind (1974) described the species as at risk of being locally extinct and recommended strict protection by keeping disturbance to a minimum. Prior to my study, there had not been any formal assessment of the NOTUGRE cheetah population.

CHAPTER 3

TRAPPING THE ELUSIVE CAT: USING INTENSIVE CAMERA TRAPPING TECHNIQUES TO ESTIMATE THE DENSITY OF CHEETAHS IN THE NORTHERN TULI GAME RESERVE, BOTSWANA

3.1 Introduction

Large carnivores play critical roles in the functioning of ecological systems and often act as umbrella species for the maintenance of biological diversity (Ferreira & Hofmeyr 2014; Ripple *et al.* 2014). Additionally, cheetahs (*Acinonyx jubatus*) and other large carnivores can be regarded as flagship species, providing revenue for eco-tourism operations (Buk & Marnewick 2010; Ferreira & Hofmeyr 2014; Ripple *et al.* 2014). However, many large African carnivores have disappeared from their historical ranges due to habitat fragmentation, prey depletion and direct persecution, and their persistence relies mostly on the success of conservation strategies (Ray *et al.* 2005). To implement conservation actions effectively it is essential to have a reliable understanding of the status of resident carnivore populations (Carbone *et al.* 2001).

Little is known about the status of cheetahs in Botswana and there is no estimate of the size of the cheetah population of the Northern Tuli Game Reserve (NOTUGRE). McKenzie (1990) vaguely refers to cheetah numbers being very low prior to 1984 but with a notable increase thereafter. Given the critical conservation status of cheetahs (IUCN 2013), it is important to have reliable population estimates to adequately evaluate the success of conservation efforts (Durant *et al.* 2007). Furthermore, the unique location of NOTUGRE, adjacent to two international borders (with South Africa and Zimbabwe), emphasizes the importance of having accurate population estimates for sound managerial decisions to be made throughout the cheetahs' range, irrespective of geopolitical boundaries.

Many aspects of cheetah ecology make it extremely difficult to monitor their populations. They occur at very low densities, and are elusive, cryptic, and highly mobile (Gros 1998; Marnewick *et al.* 2006, 2008; Durant *et al.* 2010). Cheetahs sometimes aggregate at smallscale, local transient hotspots (for example in areas with high prey densities and low predator densities), that may be miss-extrapolated to large-scale high cheetah density (Durant 1998; Durant *et al.* 2010). Additionally, the large home ranges of cheetahs may give the false impression of high cheetah numbers due to repeat sightings of the same individual(s) at several locations over large areas

(Marker *et al.* 2008a; Houser *et al.* 2009; Buk & Marnewick 2010). Direct counts of cheetahs are logistically impractical and incur high financial and time costs making them rarely feasible (Durant *et al.* 2007; Balme *et al.* 2009). Population sizes can, however, be estimated using indirect methods (Thompson *et al.* 1998; Karanth & Nichols 2002; Balme *et al.* 2009). For example, the survey of animal signs, public interviews and photograph submissions, or inferences of population densities from indices such as prey biomass and habitat suitability (Karanth *et al.* 2010).

The method selected to estimate numbers should consider the species, the area and habitat, the available budget and the amount of skilled manpower and time available. However, the objectives of the study should be of utmost importance (Karanth & Nichols 2002; Henschel & Ray 2003). The objectives of a study may vary from a simple presence-absence survey, to relative abundance, to absolute abundance and density estimates (Henschel & Ray 2003). In addition, the detection probability, which is the probability of an animal being included in the count statistic, should be considered regardless of the method such that the sample size or number of target animals detected is sufficient for sound population estimates (Nichols 1992; Karanth & Nichols 2002).

3.1.1 Camera trapping

Remotely-triggered camera trapping is a non-invasive method for monitoring rare, cryptic mammals (Carbone *et al.* 2001). It can be successfully used to systematically survey individually identifiable big cats (Soisalo & Cavalcanti 2006). Individuals can be identified by unique natural markings such as spot or stripe patterns, which allows for population estimates to be calculated by capture-recapture methods (Otis *et al.* 1978). Photographs from the surveys provide encounter history data, representing the sequence of individual observations generated from camera traps, with occasions and spatial locations of individual photo captures.

This method has been successfully used to provide population estimates for a number of individually recognizable felid species (see Table 3.1). Although camera-trapping studies of cheetahs have been completed in north-central Namibia (Marker *et al.* 2008b) and the Thabazimbi district of the Limpopo Province of South Africa (Marnewick *et al.* 2008), both studies used fewer than 13 sampling locations. O'Brien & Kinnaird (2011) also published abundance estimates of cheetahs using camera traps but the four positive returns were insufficient to derive a reliable population estimate. These are apparently the only published cheetah population estimates using camera-trapping and therefore this study represents the first intensive camera-trapping survey in Africa to estimate absolute abundance and density of cheetahs.

Table 3.1 Examples of camera trapping studies for individually recognisable felid species

Species		Studies
Tigers	*Panthera tigris*	Karanth 1995; Karanth & Nichols 1998, 2002; Carbone *et al.* 2001
Jaguars	*Panthera onca*	Silver 2004; Soisalo & Cavalcanti 2006; Kelly *et al.* 2008; Negrões *et al.* 2012; Noss *et al.* 2013
Ocelots	*Leopardus pardalis*	Trolle & Kéry 2003; Maffei *et al.* 2005, Dillon & Kelly 2007
Snow leopards	*Panthera uncia*	Jackson *et al.* 2010
Leopards	*Panthera pardus*	Henschel & Ray 2003, Balme *et al.* 2009; Gray & Prum, 2012; Borah *et al.* 2013
Pumas	*Puma concolor*	Kelly *et al.* 2008; Negrões *et al.* 2010

3.1.2 Capture-recapture method

Closed-population capture-recapture models are typically applied to estimate the relative numbers (or density) of cryptic carnivores (Nichols 1992). For the application of this method, two assumptions need to be met; 1. The population is demographically closed; and 2. Individuals cannot have zero probability of capture (White *et al.* 1982; Nichols 1992; Karanth & Nichols 2002). Population closure is practically met by using a survey period that is sufficiently short that it is unlikely that deaths, births, immigration or emigration will occur during the surveyed period (Otis *et al.* 1978; Karanth & Nichols 2002; Tobler & Powell 2013). However, the capture probabilities for cheetahs, especially in semi-arid habitats with lower prey density, are expected to be low (Gros *et al.* 1996; Marker *et al.* 2008b; Buk & Marnewick 2010).

Therefore, a balance needs to be found where survey length is short enough to satisfy population closure, but long enough for sufficient data to be collected for population estimation (Tobler & Powell 2013).

It is also essential that individuals of the target species be reliably distinguished from each other throughout the study (White *et al.* 1982). Cheetahs are individually recognisable by their unique spot patterns (Kelly *et al.* 1998) and image quality and trap placement are therefore factors which must be carefully considered (Karanth & Nichols 2002). Population density provides a useful and comparable population statistic (O'Brien & Kinnaird 2011). The density of a population is defined as the number of adult animals averaged across the study area and is typically expressed as the number of animals per 100 square kilometers (Karanth & Nichols 1998). To calculate density the effective area sampled needs to be known and is estimated by adding a buffer around the trap array (Karanth & Nichols 2002; Soisalo & Cavalcanti 2006). The effective area sampled is conventionally calculated using *ad hoc* approaches whereby the mean maximum distance moved (MMDM) or half of the mean maximum distance moved (HMMDM) by the animals being studied is calculated (Otis *et al*. 1978; White *et al.* 1982; Karanth & Nichols 1998, 2002; Soisalo & Cavalcanti 2006). This approach has been heavily criticized because the effective trapping area (ETA) varies considerably with the chosen buffer strip method, and consequently influences density estimates (Efford 2004; Soisalo & Cavalcanti 2006; Borchers & Efford 2008; Foster & Harmsen 2012; Gerber *et al.* 2012; Gopalaswamy *et al.* 2012; Noss *et al.* 2013; Tobler & Powell 2013).

A relatively novel approach has been developed using Spatial Explicit Capture-Recapture (SECR) models (Efford 2004; Borchers & Efford 2008; Royle *et al*. 2009a). SECR models incorporate the geographic locations of camera traps and the individual animal captures within the trap array, thereby accounting for unequal detection probabilities among individuals and enabling direct estimates of population size and density (Borchers & Efford 2008; Royle *et al*. 2009b; Sun *et al.* 2014). The SECR method calculates individual specific detection probabilities by estimating the activity centres of individuals and the camera trap locations (Borchers 2010; Sun *et al.* 2014). Furthermore, SECR models allow for non-regular trap locations while still providing precise estimates of abundance (Sun *et al.* 2014). SECR methods have consequently become the preferred method for calculating population estimates from camera-trapping data and have been implemented for several recent camera trap surveys of individually identifiable large carnivores (Royle *et al.* 2009a; Gardner *et al.* 2010; Kalle *et al*. 2011; Sollmann *et al.* 2011; Grant 2012; Foster & Harmsen 2012; Gerber *et al.* 2012; Mondal *et al.* 2012; Noss *et al.* 2012, 2013; Gray & Prum 2012; Tobler *et al.* 2013).

3.1.3 Objectives

In this chapter, the cheetah population density of NOTUGRE is estimated using camera trapping techniques and SECR analyses. The influence of placement of camera traps at scentmarking trees on cheetah capture rates is also investigated. Finally, the effect of survey duration on sample size and resulting population estimates for a carnivore species that occurs at a low population density is explored.

3.2 Methods

Two camera-trapping surveys were carried out using two different trapping arrays. The first survey followed the more traditional approach of having camera traps set uniformly over the landscape in a systematic pattern (Otis *et al.* 1978). The second survey had camera traps placed at sites presumed to increase the probability of capturing cheetahs, resulting in an irregular pattern of trap locations across the study area (Marker *et al*. 2008b). Both surveys were conducted in the centre of NOTUGRE on five different properties, including Mashatu, Fika Futi, Naledi, Kanda and Uitspan North, and covered approximately 240 km² (Figure 3.1). The location was chosen for practical purposes but also to avoid the edges of the reserve where theft of cameras may have been a problem.

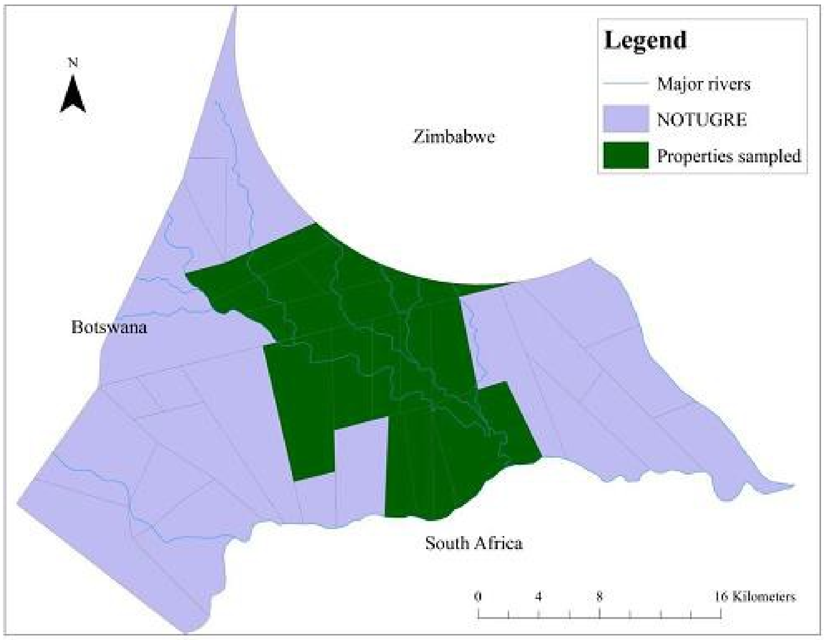
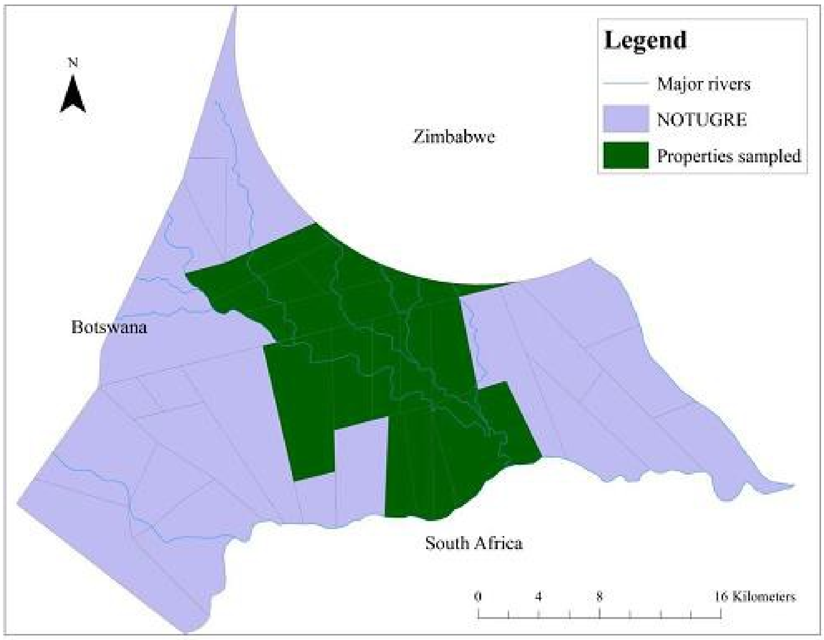


Figure 3.1 A map of NOTUGRE illustrating the properties included in the study area (green polygons) for the camera trapping surveys.



3.2.1 First Survey – Regular trap configuration

Twenty Cuddeback Attack (Non Typical, Inc., Green Bay, WI, USA) camera traps were used at 60 locations within NOTUGRE (Figure 3.2). A stratified, random sampling technique was used (Otis *et al.* 1978; White *et al.* 1982; Thompson *et al.* 1998; Borah *et al.* 2013), deploying camera traps in the best locations, typically along trails or other well-travelled animal paths (i.e. the locations most likely to capture moving animals) within a buffer. This ensured an even sampling effort across the landscape and an equal detection probability for all individuals, reducing sampling biases from spatial variation in capture probabilities (Karanth & Nichols 1998; Foster & Harmsen 2012).

Based on cheetah movements observed in the study area, a grid with equally spaced points at 3.7 km intervals was placed over a map of the surveyed area using ArcMap 10 (ESRI, Redlands, CA, USA). These predetermined points represented ideal camera trap placements, but actual camera traps were set within 200 m (mean distance and standard deviation = 164 ± 94 m) of the predetermined points, thus had a tolerance of 4.4 ± 2.5%. Camera traps were placed within this buffer zone at sites presumed to maximise the likelihood of photographing a moving animal, usually on well-defined animal paths (Balme *et al.* 2009). The 3.7 km spacing between the units was chosen to ensure that no cheetah would go undetected (Karanth & Nichols 1998, 2002). Studies designed to estimate the abundance of a species require that camera traps be placed such that the entire area sampled does not have any large gaps in which a cheetah's movements could go undetected during the sampling period (Karanth & Nichols 2002). In other words, no cheetah has a capture probability of zero. The spacing between camera traps is typically based on the average home range or minimum home range of the target species (Karanth & Nichols 2002). However, the size of cheetahs' home ranges varies substantially among geographical locations and social groups (Gros et al. 1996; Broomhall et al. 2003; Bissett & Bernard 2007) and there were no prior data on cheetah home ranges for the study area or surrounding areas. Information about the movement of a resident adult female cheetah with sub-adult cubs was available and was used to calculate the average daily distance moved. The movement data were obtained from global positioning co-ordinates taken every four hours from a satellite collar fitted to the cheetah (E. Brassine, unpublished data). The cheetah was collared by a qualified veterinarian for routine monitoring. The daily distance travelled was calculated by adding the distance between consecutive locations within a day (Hunter 1998). This average daily distance moved was used to calculate the minimum distance between camera trap sites.

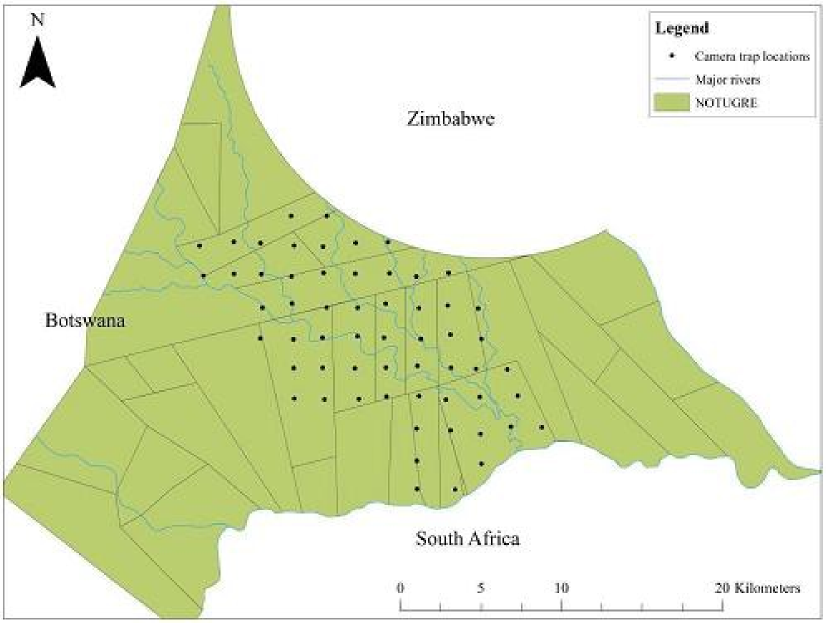


Figure 3.2 The locations of camera traps (n = 60) for the first survey using a systematic grid method. (ArcMap 10; projected: Transverse-Mercator, spheroid W GS84, central meridian 29; map units: meters).

Using two opposed cameras per station is preferable to capture both sides of the animal for individual identification and to increase the detection rate (Karanth & Nichols 2002; Negrões *et al.* 2012). However, only one camera was used per station in my survey so that more traps could be deployed over a larger area, thereby increasing the number of independent locations and maximising the chances of detecting every cheetah in the area (Foster & Harmsen 2012).

When surveying rare or sparse species it is best to sample broadly across the study area as this increases the likelihood of captures (Foster & Harmsen 2012). Typically, camera-trapping surveys are conducted over a short period to ensure demographic closure (Karanth & Nichols 2002; Royle *et al.* 2009a). My survey was carried out over a 90 day period during the hot/wet season (December – March 2013). Due to the large size of the survey area (±240 km²) and the limited number of cameras ($n = 20$), the Adjacent Block method (Karanth & Nichols 1998, 2002) was implemented to ensure that the whole sampling area was covered. The sampled area was divided into three sections and each section was sampled sequentially for approximately 30 continuous days (Karanth & Nichols 2002). Cameras were collected from their first location in one section and deployed to their new location as quickly as possible (approximately three days to move all cameras). The total number of days that cameras were active is the duration of the survey, with each day (24-h period) defined as a sampling occasion, starting at 12h00 and ending at 11h59 (Otis *et al.* 1978), when cheetahs are believed to be least active (Hayward & Slotow 2009).

The cameras were set to take high quality (5MP) images and the strobe flash range was set at 30 feet (9.14 m). This was occasionally reduced to 10 (3.04 m) or 20 feet (6.09 m) when an animal was likely to come closer to the camera so as to reduce the risk of overexposed images. The cameras used four D-cell batteries, a 4GB SD card and a passive infrared sensor to detect heat and motion. The cameras were housed in steel protective casings and fastened to trees. Chains and padlocks were also used to secure the cameras against theft. Cameras were secured approximately 0.3 m above the ground and were active 24h/day with a 1 minute delay between consecutive photographs to minimize unnecessary captures of gregarious, non- target species. The cameras were inspected, on average, every 15 days to replace batteries and memory cards and to ensure that they were operating normally. No baits were used at camera trap stations to prevent heterogeneous capture probabilities (Foster & Harmsen 2012). However, no effort was made to conceal human scent.

All data from camera traps were summarized in a comprehensive spreadsheet. The number of active days, or trap-days, was calculated for each station. Every day that a camera was active was deemed one active day. If cameras malfunctioned, had technical problems (such as no flash triggered at night or flat batteries), or were damaged by elephants (*Loxodonta africana*) or flooding, those days were excluded from the data analyses. Thus, active days included only problem-free days. Independent photographic events were defined as consecutive photographs of the same species taken more than one hour apart, or non-consecutive photographs of individuals of the same species (Tobler *et al.* 2008). The number of independent events per 100 trap days [relative abundance index (RAI)] was calculated for each species (Karanth & Nichols 2002).

3.2.2 Second survey – non-random configuration using scent-marking posts

The probability of detection is a fundamental aspect that needs to be carefully considered in order to obtain robust estimates of population size (Long *et al.* 2008). A large enough sample size relies on the capture probability of the species being studied (Otis *et al.* 1978), which depends on a number of variables, such as survey design, habitat type, prey availability, and most crucially on the behaviour of the target species (Soisalo & Cavalcanti 2006). In this study an important behavioural trait was cheetahs' communication with conspecifics through scentmarking (Eaton 1970; Marker *et al.* 2010; Soso *et al.* 2014). Scent-marking can take the form of defecation on or under a tree, urine spraying, and clawing (Eaton 1970; Marnewick *et al.* 2006; Soso *et al.* 2014). Trees are predominantly used for scent-marking posts but rocks, termite mounds and even man-made objects may also be used (Eaton 1970). Careful choice of trap location may increase the probabilities of capturing the target species, and hence produce a more accurate representation of the true population at the study site (Soisalo & Cavalcanti 2006).

The second camera trapping survey used known scent-marking posts for camera trap locations (Marnewick *et al.* 2006, 2008; Marker *et al.* 2008b). Field guides working in NOTUGRE have observed cheetahs using scent-marking posts and, with their assistance, a total of 104 such sites were identified and mapped as potential trapping locations. A proximity test was run in ArcMap 10 to calculate distances between all scent-marking posts and data were cleaned; effectively removing scent-marking posts that were within 250 m of other scent-marking posts. Where more than one scent-marking post lay within a selected area, the site with the most recent signs of cheetah activity (presence of scats, urine spray, and tracks) and with the least human interference would be selected. Accordingly, 60 camera trap placement sites were chosen (Figure 3.3); with the number of sampling points consistent with the first survey.

The furthest spacing between scent-marking posts (3.13 km) fell within the chosen required maximum distance between camera trap placements (3.7km). This ensured that there were probably no gaps sufficiently large to contain a cheetah's movements within the sampled area and that all cheetahs had a non-zero detection probability (Karanth & Nichols 2002).

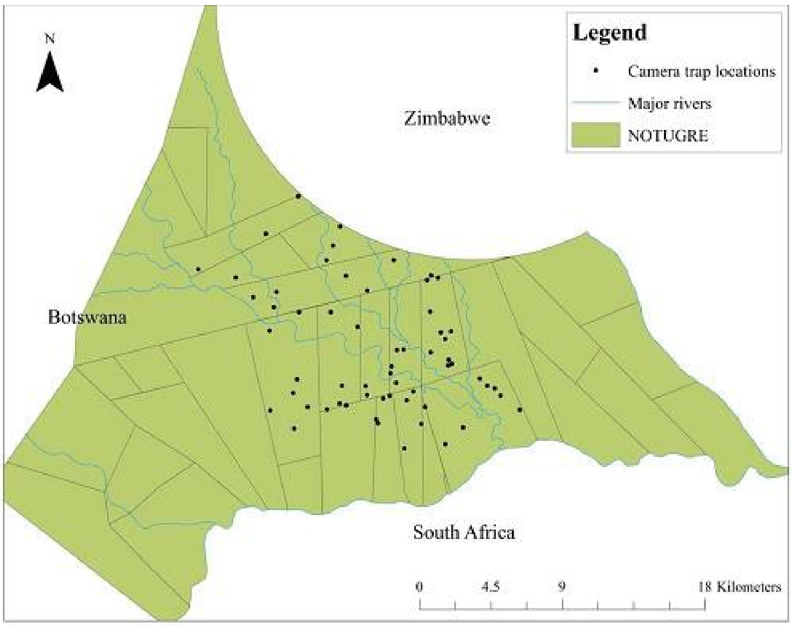


Figure 3.3 Camera trap locations (n = 60) at identified scent-marking posts for the second camera trap survey. (ArcMap 10; projected: Transverse-Mercator, spheroid WGS84, central meridian 29; map units: meters).

Cameras were set in the cheetah's anticipated path to photograph the flank of the animal because broadside images facilitate easier identification (Marnewick *et al.* 2006). Where possible, brush was packed around the scent-marking tree leaving only one access point to encourage the animals to move in front of the camera (Marnewick *et al.* 2006). A combination of Cuddeback Attack (*n* = 24) and Bushnell Trophy CamTM IR (Bushnell Outdoor Products, Overland Park, Kansas, USA) (*n* = 6) camera traps were used. Cameras were only operational at 30 locations during any given sample occasion. Thus, the Adjacent Block method was implemented, with the sampled area divided into two blocks and a camera rotation after 45 consecutive days to cover the entire sampled area. The Bushnell cameras were set to take a burst of three photographs per trigger to aid in identification, but for every trigger event, consecutive photographs were recorded as a single capture. Cuddeback Attack cameras allowed for a short video clip (30 seconds) to be taken after each daytime trigger event. This function was activated to aid individual cheetah identification. All other camera settings and positioning were as per the first camera trapping survey.

The survey ran for 90 days and was carried out during the cool/dry season (June –September 2013). Cameras were checked approximately every two weeks with an initial check after three days to ensure that the camera was operating correctly and was properly positioned to maximise the chances of captures.

3.2.3 Extended survey

Small sample sizes are typical of capture-recapture studies for carnivores that have large home ranges (O'Brien & Kinnaird 2011). Nonetheless, the dataset needs to include captures and recaptures of a sufficient proportion of the population to calculate effective sampled area and density (Foster & Harmsen 2012). A larger sample size, and hence precision, can be obtained by adapting the design of the survey, this includes using species-specific targeted placement; increasing the number of sampling points; using a larger sampling area; increasing the density of trapping points within the sampled area; and extending the duration of the sampling period (Otis *et al.* 1978; O'Brien & Kinnaird 2011).

To increase the number of captures, the survey period of the second camera trapping survey was extended after the initial 90 day survey. The 30 camera traps were left at their position for a further 40 days, extending the number of trapping days to a total of 130 days. While a long survey period may be necessary for species with low detectability to have sufficient captures for analyses (Foster & Harmsen 2012), the assumption of demographic closure may become violated (Foster & Harmsen 2012). Thus, population closure tests and SECR analyses were performed using all cheetah photographic captures over the extended sampling period. Density estimates for the two different sampling period lengths were compared using a Student's t-test.

3.2.4 Cheetah identification

Cheetah photographs were categorized and analysed with Adobe Photoshop Lightroom 3.6. All photographic captures of cheetahs were analysed by visual inspection of spot patterns to determine the identity of each cheetah and each individual was given a unique identity number (Kelly *et al.* 1998; Kelly 2001). The identification of individuals and capture events was based on the guidelines below (Caro 1994; Karanth 1995; Heilbrun *et al.* 2003).

Individuals were identified based on spot patterns or individual spots on the body, tail, legs and face. At least two, but preferably three, unique features or human-made markers (e.g. a collar) were required to identify an individual. One different feature was considered sufficient to consider that two photographs represented two different individuals. Photographs of poor quality, or where spot patterns were obscured, were marked as unidentifiable and excluded from the analysis. A photograph was considered to be a first capture if it could not be matched with any individuals in previous, older photographs. Re-captures were photographs depicting an individual already identified. All individuals were sexed based on presence/absence of scrotal testes.

The photographs were independently analysed by two people to ensure their correct classification (Kelly *et al.* 2008). If an individual's identity could not be agreed upon, these photographs were excluded from the analysis. A cheetah identikit, developed during the photographic survey (see Chapter 4 and Appendix III), was used to assist with identification. Only adult cheetahs were considered for analysis of population estimates. The sampling occasion, time, location, and individual cheetah identity of each capture event were recorded in a spreadsheet. Capture histories were prepared for each adult identified in the camera trapping survey with a sampling occasion defined as 1 day (24 hours) starting at 12h00.

3.2.5 Data analysis

Tests for population closure were performed using the CloseTest program version 3. The program tests capture-recapture data for closure using two tests (Otis *et al.* 1978; Stanley & Burnham 1999). There are a number of programs that are available for calculating population estimates using capture-recapture data such as SPACECAP and DENSITY (Efford *et al.* 2004; Gopalaswamy *et al.* 2012). These are the most commonly used programs for running SECR models, with each program using a different (and therefore independent) approach for running the analysis. SPACECAP uses a Bayesian modelling framework and DENSITY uses maximum likelihood-based approach (Gopalaswamy *et al.* 2012). Although analyses by the program SPACECAP take much longer to run than DENSITY, it was preferred as it allowed for inference about the locations of individuals that were not photographed during the survey and could thus be used for modelling demographically open populations (Gopalaswamy *et al.* 2012). Another advantage of SPACECAP is that the Bayesian framework offers non-asymptotic inferences which are applicable for small data samples typical of camera trapping studies of carnivores that occur at low densities (Gopalaswamy *et al.* 2013).

Density estimates are calculated in SPACECAP using information on capture histories in combination with the distribution of individuals (trap sites) and each traps' active days (dates when camera trap locations were active and operational), providing more accurate, precise, and hence more reliable results (Gopalaswamy *et al.* 2012). The model firstly determines an individual's activity centre and then estimates the density of these activity centres across a precisely defined area containing the trap array (Gopalaswamy *et al.* 2013). Furthermore, the models consider the traps as functioning independently and this allows individuals to be captured in multiple traps during a capture occasion and even multiple times by the same camera trap, which is realistic in camera-trapping studies (Royle *et al.* 2009a).

SPACECAP runs as a package in the program R version 3.0.2 (R Development Core Team) (Gopalaswamy *et al.* 2013). SECR analysis in SPACECAP requires specific input files.

Three input files are required; these files consist of the following:

- Animal capture detail
- Trap deployment detail
- State-space detail

Guidelines for creating the three input files can be found in the SPACECAP manual (Gopalaswamy *et al.* 2013). Spreadsheets were created using Microsoft Excel and the input files were saved in an ASCII comma separated values (.csv) format in the working directory. All X and Y-co-ordinates must be expressed in the Universal Transverse Mercator UTM projection system for computation in SPACECAP.

The third file (state-space detail) requires the creation of potential activity centres within the state-space. The state-space or 'S' represents the surveyed area containing the camera traps combined with an extended area surrounding it. The state-space is represented by a fine grid of equally-spaced points that represent all possible activity centres (or home range centres) of all of the individuals in the population surveyed (Gopalaswamy *et al.* 2013). Point spacing of 500 m is commonly used in the point array but because of the relatively large home ranges of cheetahs 1000 m point spacing was selected. The distance between points should be such that 10-20 points might lie in a single home range of an individual (A. Royle, research statistician and author of SPACECAP, pers. comm.). Potential home range centres were generated using ArcMap10 in conjunction with the Repeating Shapes for ArcGIS extension Tool (Jenness 2012). The state-space requires being sufficiently large to ensure stability in the density estimate, which usually requires a buffer strip to be added to the trap array that is two or three times larger than the encounter probability parameter (Gopalaswamy *et al.* 2013). A "Minimum Area Rectangle" is formed by connecting the outermost camera trap locations in a rectangle and a buffer is created around this minimum area rectangle.

3.2.6 Buffer

The buffer region should be sufficiently large for individual animals outside the buffered region to have zero probability of being photo-captured by camera traps during the survey. For the analysis of the data, the state-space boundaries were calculated using three different buffered distance methods (see Figure 3.4): Double the diameter of the minimum known home range size for cheetahs (11 km²; Purchase & du Toit 2000) (Buffer width = 3.74 km). The diameter of a known home range for that specific site (E. Brassine, unpublished data), approximating the home range as a circle (Buffer width = 8.97 km). The Maximum Distance Moved (MDM) (Buffer width = 28 km) – the centre point of the home range of a cheetah fitted with a satellite collar was calculated by averaging all of the GPS co-ordinates. The furthest fix from this centre point was used to measure the MDM. If home range data from more than one collared cheetah had been available, the average maximum distance moved would have been used to calculate the buffer distance, as sample size could affect this measurement.

The adequacy of each model is evaluated based on its Bayesian posterior probability (Pvalue). A model that provides an adequate description of the data will have a Bayesian Pvalue near 0.50, extreme values (near 1 or 0) indicate that the model is inadequate (Gopalaswamy *et al.* 2013).

The habitat suitability indicator column required in the third input file was created with data from Google Earth. Aerial imagery of the state-space area was used to indicate areas unsuitable for cheetahs. Selected by the author, unsuitable areas included human settlements, large water bodies, fenced agricultural farms (farms with high human activity and maintained game fences), and mining areas (Pettorelli *et al.* 2009; Gopalaswamy *et al.* 2013). Home range centres that fell on these areas were identified as locations where cheetahs could not exist and marked with a '0' next to their co-ordinates. Regions of suitable habitat were described by a grid of equally spaced points representing 1 km² over the state-space. The activity centres are assumed to be uniformly distributed over this area of suitable habitat.

The SPACECAP input files were uploaded and appropriate model combinations were chosen for analysis (Gopalaswamy *et al.* 2013). The following model definitions were selected: trap response absent, spatial capture-recapture, and detection function was set to half-normal (Gopalaswamy *et al.* 2013). The Markov-Chain Monte Carlo (MCMC) parameters were set to the recommended default values: 50 000 iterations, 1000-sample burn-in, no thinning was selected (value of 1) and data augmentation of 35 was chosen. To analyse the complete data model (the model with a fixed number of activity centres) where the number of animals in the population is unknown, the method of data augmentation is used. The data augmented must be sufficiently many that the posterior probability distribution of N is not truncated. Following the recommendation by Royle *et al.* (2009b) that the data to be augmented should be five to ten times the number of identified individuals, data augmentation was set to 35 (five times seven). The data augmentation value represents the maximum allowable number of possible animals within the state-space (Gopalaswamy *et al.* 2012). The behavioural response was not chosen as baits or lures were not used in the survey, thus an individual's encounter probability before and after the initial encounter was expected to be similar. Movement of individuals was non-random in this case as individuals will use certain scent marking posts within their home range (Caro 1994).

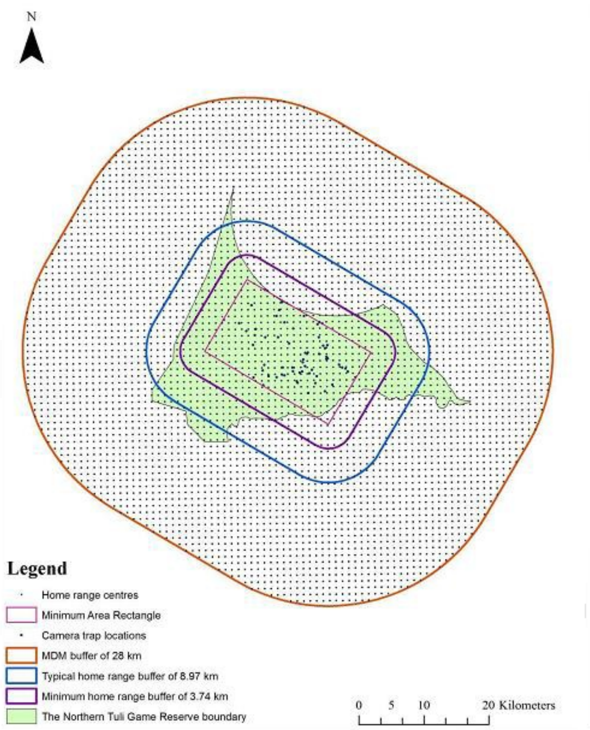


Figure 3.4 An example of the spatial data created in ArcMap 10 for the third input file "Potential Home Range Centres" for the program SPACECAP showing the state-space boundaries for three different buffered distances.

3.3 Results

3.3.1 First Survey – Regular trap configuration

A total of 1616 active days were logged during which 3346 animal photographs were taken and only nine (0.27%) were photographs of cheetahs. Cheetah photographs were recorded at only two of the 60 sampling locations and all but one of these events occurred at a camera trap station that had been placed at a known cheetah scent-marking post. From the photographs 32 mammal species and 23 bird species were identified and no reptile species were captured. Relative abundance indices (RAI) and the proportion of total photographs taken are shown in Appendix II for all recorded mammal species. The most common mammal species, based on capture frequencies (CF > 2.0), were impala (*Aepyceros melampus*), followed by elephant, giraffe (*Giraffa camelopardalis*) and eland (*Tragelaphus oryx*). Eleven predator species were identified and the most frequently photographed were spotted hyena (*Crocuta crocuta*), blackbacked jackal (*Canis mesomelas*), and leopard. The least common species were bushpig (*Potomachoerus porcus*), lion (*Panthera leo*), banded mongoose (*Mungos mungo*), and bushbuck (*Tragelaphus scriptus*) (all photographed only once). No further analyses to assess cheetah population size were carried out due to the insufficient number of cheetah captures.

3.3.2 Second survey – non-random configuration using scent-marking posts

The second study had a total of 2660 active camera trapping days and of the 3323 animal photographs, 53 (1.6%) were of cheetahs captured at 11 of the 60 camera trap sampling locations. Forty-nine species, including 28 mammal and 21 bird species, were recorded. Appendix III shows the RAI and proportion of total photographs taken for the entire mammal species recorded. This survey detected two mammal and nine bird species which were not recorded in the first survey. However, five mammal and 11 bird species which were captured in the first survey were not recorded by the second survey. Anthropogenic activity was high (4.4%; n = 116 photographs), because many of the scent trees were placed on hills used as stopping points during game viewing drives.

Cheetah photographs made up 18 independent capture events. A capture event includes all photographs of an individual within a 24 hour activity period at a camera station (O'Brien *et al.* 2003; Gerber *et al.* 2012). A total of seven adult cheetahs were identified from photographs (two females; five males) and five cheetah photographs were excluded from the analysis as the individuals could not be identified. However, the cheetahs in these photos were cubs and would have been excluded from the analysis regardless of whether identification was possible or not. Details of individual cheetah visits, capture location and capture occasion are shown in Table 3.2.

Table 3.2 All capture and re-capture details of individual cheetah visits (sample occasion) recorded from the second camera trapping survey in the Northern Tuli Game Reserve, Botswana. Independent capture events used for analyses are not shown.

Sample Occasion*	Time	Location ID	Number of photographs	Cheetah ID
9	11:46	24	1	CM6
10	12:07	25	1	CM5; CM6
38	10:49	25	1	CM3
45	11:14	18	1	CM2
46	14:31	22	2	CM1; CM2
46	15:48	22	4	CM1; CM2; unidentifiable 50
50	13:20	58	3	CF3; 2 cubs
63	06:33	54	3	CF3; 3 cubs
72	06:33	56	5	Cub
78	05:18	33	3	CM5; CM6; unidentifiable 85
87	10:43	40	6	CF4
87	19:34	58	2	CM2; CM3
87	04:46	58	3	CM1
87	04:51	58	2	CM2
87	07:28	57	1	CM1; CM2; CM3
88	07:45	56	2	CM2; CM3
88	20:20	55	9	CM1; CM2; CM3
88	05:09	55	1	CM1
88	05:26	54	3	CM1; CM2; CM3

*Sample occasion refers to the day on which cheetahs were captured within the survey period with sampling occasion 1 referring to the first day of sampling.

Capture frequency ranged from one to five per individual, with an average of 2.86 captures per individual. The number of photographs per sampling occasion ranged from one to nine, with an average of 2.79 per sampling occasion. Latency or time delay to first photograph for each individual ranged from 9 to 85 days.

3.3.3 Extended survey

A further 1090 days were logged from the extended survey period which resulted in a total of 3750 recorded active camera trapping days (Table 3.3). An additional seven camera trap locations photo-captured cheetahs, which accounted for a total of 13 events including 147 cheetah photographs, increasing the total sample size from 18 to 31 capture events at a total of 18 camera trapping locations (Table 3.3). No new individual cheetahs were recorded during this extended survey. However, capture frequency ranged from two to 10 per individual, with an average of 5.57 captures per individual. Capture details of individual cheetah visits are shown in Table 3.4.

Table 3.3 Summary of the first, second and extended camera trapping surveys conducted in NOTUGRE.

	First survey	**Second survey**	**Extended survey**
No. of active camera-trapping days	1616	2660	3750
Total number of photo-captures	3346	3323	4823
Cheetah photo-captures	9	53	200
Cheetah capture events	5	18	31
Number of individual cheetah identified	2	7	7
Number of sampling locations that captured cheetahs	2	11	18
Capture frequency per individual (mean)	N/A	2.86	5.57

Table 3.4 Capture details of individual cheetah visits (sample occasion) recorded during the extended second camera trapping survey in the Northern Tuli Game Reserve, Botswana.

Sample occasion	**Time**	**Location ID**	**Number of photographs**	**Cheetah ID**
101	1:58	33	2	CM6
101	9:13	44	32	CM5; CM6
124	11:41	51	1	CF4
126	6:20	47	3	CF8 (cub)
130	2:37	52	3	CM1; CM2; CM3
130	3:16	51	2	CM2; unidentifiable
130	3:36	33	2	CM6; unidentifiable
130	3:49	47	3	CM2; unidentifiable
130	7:43	41	9	CM2; CM3
131	23:03	43	1	CM5
131	3:42	38	3	CM1; CM2; CM3
133	19:18	33	1	CM5; CM6

3.3.4 SECR analysis for the second camera trapping survey

Closure tests were inconclusive due to the small dataset that only had a few individual captures and recaptures. Small sample sizes and unequal capture probabilities can negatively affect closure tests (Otis *et al*. 1978; Soisalo & Cavalcanti 2006). However, previous studies of large felids have indicated that a three-month sampling period is sufficient to meet the closure assumption (Karanth 1995; Karanth & Nichols 1998). Density estimates were sensitive to the buffer width estimator, with density estimates increasing with a decreasing state-space area (Table 3.5). Cheetah density over the state-space with a buffer of 28km was estimated at 0.55 cheetahs/100 km² with a 95% confidence interval of 0.24 (lower level) and 0.90 (upper level). However, a buffer width of 8.97 km showed considerable difference with an estimated density of 1.24 cheetahs/100km² and the smallest buffered distance used (3.74km) for the state space area estimated cheetah density at its highest with estimates of 1.69 cheetahs/100km². Bayesian P-values for the different models suggest that all models may be appropriate (Table 3.5).

Table 3.5 Summaries of the Bayesian SECR analyses using three different buffer width estimators to create the state space area. The Bayesian P-value gives the adequacy of each model.

Buffer Width (km)	**Variables***	**Mean**	**SD**	**95% Lower HPD level**	**95% Upper HPD Level**	**Bayesian P-value**
28 km buffer; state space area of 4708 km²	sigma	6.17	1.63	3.57	9.60	0.46
	lam0	0.02	0.01	0.00	0.04	
	Psi	0.60	0.22	0.24	1.00	
	Nsuper	25.39	9.09	11.00	42.00	
	Density	0.54	0.19	0.24	0.90	
8.97 km buffer; state space of 1166 km²	sigma	4.98	1.37	2.78	7.59	0.56
	lam0	0.01	0.00	0.00	0.02	
	Psi	0.35	0.16	0.10	0.67	
	Nsuper	14.22	6.27	7.00	27.00	
	Density	1.24	0.55	0.61	2.36	
3.74 km buffer; state space of 591 km²	sigma	4.79	1.12	2.90	6.96	0.58
	lam0	0.00	0.00	0.00	0.01	
	Psi	0.25	0.09	0.09	0.43	
	Nsuper	9.90	2.72	7.00	15.00	
	Density	1.69	0.46	1.19	2.56	

* Sigma (σ) represents the range parameter of an animal; lam0 (λ_0) is the expected encounter rate and can be used to estimate capture probability; parameter psi (ψ) represents the proportion of the actual number of animals and the maximum allowable number which was set during data augmentation; N_{super} is the population size of individuals for the prescribed state-space; density is calculated from the estimated number of activity centres located in the state-space and is expressed as individuals per 100 km².

3.3.5 Model refinement

Following recommendations by A. Royle (pers. comm.), the author of SPACECAP, the MCMC parameters were changed to the following: number of iterations 100 000 and burn-in values of 2000 generations to accommodate for the small sample size and large movements observed in the dataset. Buffer widths of 20, 25 and 30 km were tested and posterior summaries were used to estimate when the density estimates would stabilize. Based on this approach, the 28 km buffer, forming a 4708 km² state-space was considered to be the most appropriate method for calculating the buffer width. Another SECR analysis was run in SPACECAP using this buffer width with 200 000 iterations and 4000 burn-in generations as MCMC parameters, higher MCMC parameters were chosen to ensure that the key parameters mixed well and reached stability. Table 3.6 presents the results (posterior mean, posterior standard deviation and 95% confidence limits) for the model parameters. A Bayesian P-value of 0.50 was calculated, which represents the best fit model and therefore suggests that this model represents the most parsimonious cheetah population density estimate. The model estimated 0.61 ± 0.18 adult cheetahs per 100 km² with a 95% maximum of 0.9 and minimum of 0.3 cheetahs/100km². An absolute abundance of 4 cheetahs (range: 2 - 6 individuals) was estimated for the ~700 km² reserve. However, NOTUGRE is probably too small to contain the home ranges of all resident cheetahs and it is therefore likely that the absolute abundance at any one time may be higher.

Table 3.6 Density estimates of cheetah using MDM buffer width of 28 km and MCMC parameters set at 200 000 iterations and 4000 burn-in generations. Density is expressed as the number of cheetahs per 100 square kilometres.

Variables	**Mean**	**SD**	**95% Lower HPD level**	**95% Upper HPD Level**	**Bayesian posterior probability**
sigma	5.10	1.02	3.27	7.14	0.5
lam0	0.03	0.03	0.01	0.07	
Psi	0.67	0.20	0.32	1.00	
Nsuper	28.56	8.38	14.00	42.00	
Density	0.61	0.18	0.30	0.90	

3.3.6 SECR analyses for the extended survey

The population closure test was again inconclusive due to insufficient data. The SECR population estimate for the 130-day period produced a density estimate of 0.58 ± 2.0 adult cheetahs/100 km² (abundance of 4 cheetahs; range 1 – 6 individuals), using the same MCMC parameters that were used for the refined model (Table 3.7). This estimate is slightly lower than the density estimated in the 90 day survey (0.61 ± 0.18 cheetahs/100 km²) (Table 3.6), but the population means did not differ significantly (t-test; t = 1.22; df =226; $p > 0.05$). Summaries of the extended survey are shown in Table 3.7.

Table 3.7 Density estimates calculated from capture histories of the extended survey using the MDM buffer width of 28 km and MCMC parameters set at 200 000 iterations and 4000 burn-in generations. Density is expressed as the number of cheetahs per 100 km².

Variables	**Mean**	**SD**	**95% Lower HPD level**	**95% Upper HPD Level**	**Bayesian P-value**
sigma	6.29	1.57	3.78	9.48	0.49
lam0	0.01	0.01	0.00	0.02	
Psi	0.64	0.22	0.25	1.00	
Nsuper	27.10	9.23	11.00	42.00	
Density	0.58	0.20	0.24	0.90	

3.4 Discussion

When comparing methods, the level of confidence in the results is usually described as precision, with high precision referring to relative certainty in the estimation of parameters (Long *et al.*

2008). To have estimates as close to actual numbers as possible (and recognising that these fluctuate) it is important to minimise bias and this is achieved in the design of the survey (Long *et al.* 2008). The design itself relies heavily on the biogeographic characteristics of the species and the objective of the survey. Estimating abundances and densities of rare species requires more effort as detection rates will be much lower and this must be taken into account. Furthermore, certain sampling methods may be effective only for particular species; this is mostly due to habitat characteristics that strongly influence animal movements and therefore rates of encounter (Karanth & Nichols 2002; Long *et al.* 2008). Species that are found in dense bush may be forced to move along natural trails; so placing camera traps on these well-defined paths accounts for such habitat heterogeneity (Karanth & Nichols 1998; Henschel & Ray 2003; Long *et al.* 2008). However, it may be more difficult to predict movements of species, such as cheetah, that occur in more open landscapes. Typical camera trapping surveys have camera trap stations set out systematically across the landscape (i.e. the first survey) and camera traps are placed along animal trails, however, cheetah capture was low with the only valid captures taken from a camera trap set at a known scent marking post. The low photo-capture rate of the cheetahs was attributed to low detectability. Given that cheetahs occur at low population densities (Caro 1994) and the unpredictable nature of their movements, the location and placement of camera traps is a critical component to a successful camera trapping survey (Karanth & Nichols 2002, Blake & Mosquera 2014). The design of the survey should have camera traps placed to maximize capture probability (Karanth & Nichols 2002; Henschel & Ray 2003). The second survey had camera traps located at cheetah scent-marking posts identified by local guides. Scent-marking posts provided ideal set up locations as they were frequently utilised by cheetahs. In addition to high probabilities of cheetah captures, cheetahs would stay at the scent-marking post long enough to obtain clear photographs; often sniffing the tree and scent-marking for a few minutes before moving on. This would not only give the camera the chance to capture the subject moving but also often resulted in multiple photos of an individual during a single capture event, sometimes providing a full individual profile (i.e. left- and right-hand side photographs). This was also noted by Marnewick *et al.* (2006). A possible drawback to such a camera trapping survey is the possible variation in individual detectability, particularly in relation to age, sex and dominance (Otis *et al*. 1978). It has been observed that female cheetahs may use scent-marking posts less frequently than males and this difference in detection probability may bias estimates and under-estimate population abundance (Marker 2002; Marnewick *et al.* 2006; Marker *et al.* 2008b). Female cheetahs rarely scent-mark, unless they are in oestrous (Marnewick *et al.* 2006). However, in the second survey two females were photo-captured at three scent-marking trees on three different occasions. These females were not believed to be in oestrous at the time (pers. obs.). Although males use scent marking posts more frequently than females (Bothma & Walker 1999; Marnewick *et al.* 2006), provided that sufficient devices are used and the study is carried out over a sufficiently long survey period, this bias should have a minimal effect on the results. Alternatively, where sample size permits these sources of heterogeneity can be addressed by including sex-specific encounter rates but this survey did not have sufficient recaptures to allow such stratification (O'Connell *et al.* 2011).

A capture-recapture study requires a relatively large number of recaptures to produce precise results (Otis *et al.* 1978; Long *et al.* 2008). However, sample size is affected by the size of the sampled area, the number of camera traps used, and the number of trapping occasions and, most importantly, on capture probability (Otis *et al.* 1978). Sampling effort can be controlled through the size of the sampled area and the number of camera traps used (Karanth 1995). It is traditionally recommended that the surveyed area be at least four times the size of the average home range of the target species (Otis *et al.* 1978), but this is logistically and financially impractical for a wide range of vertebrate species (Foster & Harmsen 2012).

Alternatively, the duration of the survey can be extended judiciously. Ninety days is the recommended maximum number of days to maintain the population closure assumption when studying large felids (Karanth & Nichols 2002). However, when surveying for species that occur at very low population densities, such as cheetahs (Caro 1994) this recommended maximum number of trapping occasions may be insufficient due to the small sample size and high latency to first detection which may be due to the cheetah's large home range. Lengthening the sampling duration beyond this maximum may improve the robustness of the results but requires careful consideration of the temporal closure assumption (Foster & Harmsen 2012). Nonetheless, increasing the length of the survey may be appropriate for some species with long life expectancies and to areas with long seasons (O'Brien & Kinnaird 2011). Furthermore, SPACECAP can calculate the density of demographically open populations (Gopalaswamy *et al.* 2012). Extending the total number of sampling occasions in this study provided substantially more photographs and independent captures, increasing the degree of certainty in the associated density (O'Brien & Kinnaird 2011). However, no new individuals were caught and a larger sample size appeared to change the density estimate little, suggesting that a 90-day survey provided sufficient data for a robust population estimate. The spatial scale of the study area (±240 km^2) may have been insufficient to incorporate sufficient home ranges of cheetahs, thereby rendering the population closure test inefficient (Otis *et al.* 1978).

The classical likelihood-based capture-recapture (CR) methods are often preferred due to their simple formulae and procedures for carrying inference, including calculating standard errors, model selection by Akaike's Information Criterion and assessing goodness-of-fit (Otis *et al.* 1978; Karanth & Nichols 1998; Royle *et al.* 2009a). However, it is difficult to assess the validity of these procedures particularly when using small sample sizes (Royle *et al.* 2009a; Gerber *et al.* 2012). Programs using SECR models such as SPACECAP are believed to be more robust than conventional CR methods and are thus considered more reliable (Foster & Harmsen 2012; Gopalaswamy *et al.* 2012). Among a number of advantages, spatial models allow for heterogonous detection probabilities among individuals and the Bayesian approach accommodates for small sample sizes typical of camera trap surveys for low density species

Density estimates calculated using different buffer widths showed considerable differences in this study. Density estimates will be overestimated if the state-space area is too small. To overcome this sensitivity, the data were analysed using different buffer widths until the values of the variables stabilised and models gave an adequate Bayesian P-value. MCMC parameters also need to be carefully considered to insure that MCMC chains reach stationarity (Noss *et al.* 2012). Methods of calculating the buffer width are clearly important when estimating density. The buffer width used should encompass the maximum movements of individuals caught on camera traps (Otis *et al.* 1978; Balme *et al.* 2009). However, the conventional MMDM method using capture location data of cheetahs from camera traps was avoided as the surveyed area was likely to be smaller than the average home range of a cheetah in the study. A cheetah fitted with a satellite collar (CF3) photographed in this survey had a home range that expanded beyond the entire survey width (E. Brassine, unpublished data; Figure 3.5). The MDM buffer width was therefore used to encompass all possible large home ranges. Nonetheless, it should be noted that the variation in calculated densities may also be an artefact of the small dataset with limited recaptures, as small sample size and insufficient recaptures are known to compromise the robustness of the analyses (Otis *et al.* 1978). Although it is important to have a standardised approach when designing a camera trapping survey and calculating buffer distances, it is equally important to understand the limitations of individual surveys and adapt the method to report population size estimates as accurately as possible. However, the methods and the justification for their use should be reported in detail so that the study can be adequately repeated and compared across different study populations.

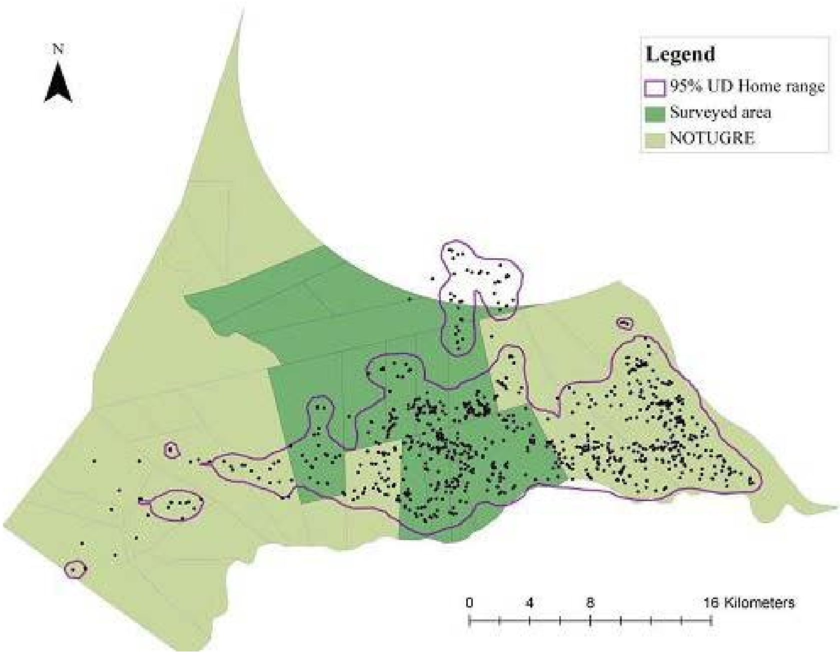


Figure 3.5 95% Kernel UD home range estimates for the collared cheetah (CF3) with GPS fixes used in the home range analysis. The camera trapping surveyed area is also presented.

Although SPACECAP takes into account unsuitable habitats, it is left to authors to interpret this as they see fit. Hence, in this study, only areas with high human disturbance and large water bodies were considered unsuitable and excluded. Elevation, vegetation type, prey availability and competitor avoidance are all likely to affect habitat suitability and hence cheetah densities (Ray *et al.* 2005; Pettorelli *et al.* 2009). For instance, cheetahs avoid lions and an inverse relationship in their densities has been documented (Durant *et al.* 2004). Also, prey density outside the protected area is likely to be low and human encounters (e.g. with poachers) are not necessarily restricted to human habitations, which may influence the expected density of cheetahs (Pettorelli *et al.* 2009). Another drawback to the method is its inability to incorporate captures of unidentifiable individuals. Nonetheless, the program incorporates aspects of trap location, active days and capture locations and thus makes it a more powerful tool than the CAPTURE program using conventional capture-recapture models and the MMDM approach.

3.5 Conclusion

Obtaining reliable population estimates for cheetahs is particularly challenging. This study is the most intensive study of cheetahs using camera traps and SECR analysis to date. Furthermore, the results provide the first population estimate for cheetahs in NOTUGRE.

This study demonstrates that using species-specific targeted placement and increasing the number of trapping days for low population densities can produce larger sample sizes for more reliable density estimates and thereby increasing the precision of the results. Camera trapping for cheetahs needs to be performed over a large area, over a long survey period and at cheetah-specific sampling points to calculate robust density estimates. I would recommend integrating multiple survey methods when assessing population sizes as this can contribute additional information and cross-validate the quality of the results (Long *et al.* 2008). In this study, photographs from the photographic survey (see Chapter 4) were used to construct individual cheetah profiles and aid in the identification of cheetahs captured in the camera trapping survey. The method can easily be replicated to perform long-term population monitoring, it is non-invasive and requires minimum personnel in the field. Finally, density results from within the reserve should not be used to extrapolate density outside the study area as there are differences in vegetation cover, prey density, human activity, land-use, and density of other large carnivores.

CHAPTER 4

CITIZEN SCIENCE IN CHEETAH RESEARCH: ESTABLISHING POPULATION ESTIMATES AND SPACE-USE OF CHEETAHS BY WAY OF A PHOTOGRAPHIC SURVEY

4.1 Introduction

The considerable global population decline and range contraction of the cheetah (*Acinonyx jubatus*) is well documented and is attributed predominantly to prey depletion, habitat degradation and conflict with humans (Marker 2002; Ray *et al.* 2005; Marker *et al.* 2010). Conservation management of cheetahs therefore relies on reliable assessments of the status of individual populations and the drivers of population trends.

The wealth of information on cheetah ecology, including behaviour, reproduction, ranging patterns and ecological requirements has primarily been generated from long-term studies in the Serengeti National Park, Tanzania (Caro 1994; Kelly *et al.* 1998; Kelly & Durant 2000; Kelly 2001; Durant *et al.* 2007), and more recently from studies undertaken in Namibia (Marker 1998, 2000; Marker *et al.* 2003b, 2008a). A key benefit to long-term studies is that they supply vital information on population trends and demographic parameters of a population, these are important in understanding population dynamics and viability, information that is crucial for the conservation management of cheetahs (Durant *et al.* 2007; Durant *et al.* 2010; Marnewick & Davies-Mostert 2012).

Research on the cheetah populations of Botswana has been limited, despite Botswana being considered a stronghold for the species in southern Africa (Bashir *et al.* 2004; Ray *et al.* 2005; Purchase *et al.* 2007). Unfortunately, the long-term studies that are required to better understand the status of cheetahs in Botswana do not exist, and population viability analyses require many years of research involving significant financial and human resources (Durant *et al.* 2007). However, there are a number of alternative techniques for collecting meaningful information on cheetah population status and distribution over a relatively short period (Bashir *et al.* 2004).

Cheetahs occur at low population densities and have large home ranges (Caro 1994) so the probability of locating individuals in the wild is low, making them difficult to survey (Marnewick & Davies-Mostert 2012). Cheetahs are individually recognisable from their unique spot patterns (Durant *et al.* 2007; Kelly *et al.* 2008) so reliable population parameters, including abundance and demographics, can be determined by reliable identification of individuals. Drawing on incidental sighting records, sightings and photographs of cheetahs taken by the general public can be a useful method for monitoring cheetah population trends for well-visited areas and areas that have habituated cheetahs (Kemp & Mills 2005; Durant *et al.* 2007; Marnewick & Davies-Mostert 2012). Photographic survey is a method used for estimating wildlife abundance and demographics of individually recognisable species by using people already in the field (citizen scientists). Such an approach reduces labour costs and can be conducted over large spatial scales and short time spans, which may make estimation of low density species possible where other methods have proven ineffective (Gros *et al.* 1996). Furthermore, the method indirectly generates public awareness of conservation issues, which may create a sense of stewardship (Marnewick & Davies-Mostert 2012).

Photographic censuses of cheetahs have been conducted in the Kruger National park (KNP), South Africa (Bowland & Mills 1994; Kemp & Mills 2005; Marnewick & Davies-Mostert 2012; Marnewick *et al.* 2014), the Kgalagadi Transfrontier Park, South Africa (Knight 1999) and the Timbavati Private Nature Reserve adjacent to KNP (Dyer 2013). National Parks have high numbers of visitors allowing for high search effort (Marnewick *et al.* 2014), while commercial private game reserves offer high-end, exclusive game viewing and usually have fewer visitors than National Parks. Non-commercial private reserves are generally visited even less frequently. Thus, conducting a photographic survey on a privately owned game reserve may be challenging, but game viewing on most private reserves is not limited to roads, potentially increasing the detectability of cheetahs. Furthermore, owners, shareholders, guides and returning visitors may provide a source of both recent and older photographic records of cheetahs within their reserves; a wealth of information that would otherwise not have been collected. Historical sightings with photographic records may therefore be used for temporal comparisons and for estimating age and family relations of known individuals.

The Northern Tuli Game Reserve (NOTUGRE), in Botswana consists of 36 privately owned properties; some are used for commercial ecotourism and others on a more exclusive, private basis. Little is known about the status of cheetahs in NOTUGRE and because cheetahs occur at low population densities and have large space requirements, they may range beyond its borders (Durant *et al.* 2007). Farmlands outside of protected areas may provide refuges from dominant predators such as lions (*Panthera leo*) (Purchase *et al.* 2007; Marker *et al.* 2010), but increase the risk of threats from humans such as direct persecution from livestock farmers (Marker 1998). Areas outside protected reserves, therefore, are important for the survival of cheetahs and the current distribution and movement of cheetahs both within and outside formally protected areas are hence vital components for the conservation of cheetahs.

In this chapter I explore the suitability and effectiveness of a photographic survey in a private game reserve to estimate the minimum population size and status of the cheetahs of NOTUGRE. In addition, the GPS locations of recognisable cheetahs allowed for a preliminary assessment of cheetah distribution, home range size and the possible movement of cheetahs across international boundaries.

4.2 Methods

4.2.1 Data collection

Between January 2012 and November 2013, tourists, staff members, shareholders and neighbouring residents were asked to submit photographs and details (e.g. location, date and time of sighting, group number and sexes) of any cheetah sightings within or adjacent to NOTUGRE. At each tourist lodge, experienced field guides took tourists on wildlife-viewing game drives twice daily, tracking and locating sought-after species, including cheetahs. Digital cameras (n = 4) (either a Canon PowerShot SX260HS or a Nikon Coolpix S9300) with built-in GPSs were given to field guides who were asked to photograph any cheetahs seen on such drives. In addition, on most days during the study period, I would drive out into the reserve, actively looking for cheetahs and following up on reports of sightings. At each sighting, the primary aim was to obtain clear photographs of the left and right flanks of every individual. As many other photographs as possible, showing different positions, were also taken to build a complete individual profile for each cheetah (Maddock & Mills 1994).

Cheetah photographs generated from the camera trapping survey (see Chapter 3) were also used to supplement the dataset. Moreover a number of residents had camera traps set out for either recreational or research purposes which captured cheetahs. All cheetah images from camera traps were included in the dataset.

This photographic survey was promoted through the distribution of pamphlets at all 17 camps within NOTUGRE (Figure 4.1). All staff and shareholders of the properties were informed of the survey through email and personal contact. Furthermore, I was based at the largest commercial lodge (Mashatu Main Camp) and actively promoted and encouraged visitors and staff to submit their cheetah photographs and sightings information. This approach also enabled visitors to learn more about the conservation of cheetahs and the cheetah research taking place on the reserve.

The survey was advertised on the Mashatu Game Reserve website (www.mashatu.com) and several popular blogs (e.g. www.blog.mashatu.com). In addition, appeals for cheetah photographs were made on a local news website, DUMELANG (www.dumelangmusina.co.za), and appeals were also made at the local Greater Mapungubwe Network meetings (Minutes from quarterly meetings 2012, 2013).

Figure 4.1 An example of the pamphlet (front and back) distributed to all the camps in NOTUGRE to create awareness of the cheetah photographic census.

4.2.2 Data collation

Digital photographs were received via email and by hand after approaching visitors and reserve managers at the various lodges. Historical photographic sightings dated back to January 2006 and included data until November 2013, with the majority of sightings received in 2013.

Each sighting was entered into a Microsoft Excel spreadsheet and included all available biological information on the sighting (viz. Cheetah ID, group name, time, date, season, number of individuals, age classes, sexes, location, activity and prey item if seen feeding) and details of the photographer, including name, contact details and the number of photographs per sighting.

Sightings were sorted by sighting date. To avoid autocorrelation of sightings, sightings of the same individual or group of cheetahs on the same morning or afternoon were pooled into one sighting event such that a maximum of two sightings per individual or group were recorded per day (i.e. morning and afternoon). All photographs were stored on an external hard drive (Transcend StoreJet 1TB) in folders identified by the photographer's name.

The program Adobe Photoshop Lightroom 5.0 was used to manage all photographic data. Lightroom is an image data management program that allows the user to organise and catalogue large numbers of photographs. Furthermore, the program automatically reads the metadata of the photographs, such as the date and time of capture, and allows for tagging with specific keywords, making it easier to sort the image database. All photographs were organised into virtual 'Photo Collections' according to each individual cheetah; whereby all photographs of the same individual were grouped into one virtual folder. The program makes a virtual copy of the photographs in these folders such that the original photographs are not moved from their original locations. All photographs were tagged with all relevant information including group size, sex and the cheetah's identity number which allowed filtering of these specific criteria and simplified searching for matches. The software also provides a secondary window display which was used to compare photographs to aid in identifying individuals.

4.2.3 Individual identification

All photographs were examined by eye and individuals were identified based on unique spot patterns following the method described in Chapter 3. Each cheetah was assigned a unique identity number consisting of two letters and a number. The first letter referred to the species (C = Cheetah), the second letter, the sex (F = Female; M = Male; US = Unknown sex). The number referred to the position in the identification sequence. Individual profiles were created for each cheetah; if only one side of the cheetah was available, half profiles were created. A cheetah identikit of all identified individuals was created for reference purposes. (Appendix IV).

Some sightings could not be used for population estimation because they either did not have photographic records or were accompanied by photographs from which the cheetah could not be identified. However, both of these types of sightings could be used for distribution mapping to display the overall occurrence of cheetahs across the landscape. Some photographs were submitted with no supporting data. In these instances, the individuals in the photographs were identified but not used for population estimation or distribution mapping.

4.2.4 Long term trends and population demography

Seasons were categorized based on the amount of rainfall distributed over the year on a monthly basis, using the rainfall records between 1996 and 2013 from three different locations within NOTUGRE. Two main seasons are experienced, the drought or dry season and the wet season (Balinsky 1962). The dry season is the period during which there is typically less than 10 mm of rainfall per month for at least three consecutive months (Balinsky 1962). During the dry season, trees lose their leaves, grass dries up and becomes exposed. The wet season is defined as the months where the bulk of the annual rainfall is received (±90% annual rainfall over a 6 month period). The two seasons are accompanied by changing average maximum and minimum temperatures. The dry season has lower average minimum and maximum temperatures than the wet season. Accordingly, the cool dry season was defined as the months of May to October (monthly rainfall: 7.2mm; monthly maximum temperature: 27.6°C; monthly minimum temperature: 12.7°C) and the hot wet season was defined as the period from November to April (monthly rainfall: 59.3mm; monthly maximum temperature: 32.7°C; monthly minimum temperature: 21.5°C). Seasonal differences in the total number of cheetah sightings were calculated for the entire survey period (Jan 2006 – Nov 2013).

The collection of photographic records was used to estimate population numbers, minimum estimated age and family relatedness. To assess general population trends the number of individuals (regardless of the frequency of sightings) was summed for each year. Individuals that had probably died were subtracted from this total. The minimum estimated ages for individuals seen continuously over more than one year were calculated at the end of 2013. The number of times an individual cheetah or cheetah group was sighted varied. Thus, the minimum estimated age and family relatedness could only be determined for individuals that had a sufficient number of repeat sightings and were seen regularly (≥ 5 sightings) for at least a year (n = 16 individuals). Photographs of cubs provided a means to determine relatedness (Kelly 2001) and approximate ages of individuals in the population without the use of genealogical records. The approximate date of birth of a cheetah was estimated as precisely as possible using the method described by Caro (1994); if the cheetah was first sighted as an adult, it was assumed to have been born at least two years prior to the first sighting (Caro 1994), therefore estimated ages for adult animals are to be considered minimum values (Kelly *et al.* 1998). Cubs were aged by comparing their body sizes against that of known-aged cheetahs using the aging scale described by Caro (1994). The first sighting of individuals was used to back-date the approximate date of birth (Caro 1994).

4.2.5 Photographic survey - Minimum population size (NOTUGRE)

The minimum population size was estimated following the methods of previous cheetah photographic censuses conducted in the KNP (Kemp & Mills 2005; Marnewick & Davies-Mostert 2012, Marnewick *et al.* 2014). The analysis for the minimum population size for NOTUGRE included all photographic sightings within NOTUGRE over a seven-month period (April – October 2013) that was selected because it included the greatest number of sightings (22.7% of all sightings and 52.2% of all submitted photographs) and overlapped with the camera trapping survey (see Chapter 3). Thus, all photographs, including those taken by camera traps, could be included in the analysis. All cheetahs that were realistically alive on the 1st of July 2013 were included in the estimate; this included all cheetahs that had been recorded for the three months prior to this date (April - June) and all adults and sub-adult (> 3 months old) cheetahs that were sighted during the four months after this date (July - October) as they would still have been alive during the census months (Marnewick *et al.* 2014).

4.2.6 Distribution and home range

All cheetah locations (n = 395) from sightings between 2006 and 2013 were used to asses space use. Locations were acquired from direct observations using global positioning systems (GPS, Garmin GPSMAP 62) or by using the closest known landmark if a GPS location was not available. Some sighting reports were not accompanied with accurate locations; in these cases, if the name of the property was known, a central location on the property was recorded (Watermeyer 2012). The location of the sighting was given a rating based on accuracy. A *1* was given if it was the exact location (within 50 m), a *2* was given if the location was of the closest prominently-known landmark (within 2 km) and a *3* if the location was inaccurate (> 2 km). The distribution of all sightings was mapped and kernel utilization distribution (UD) method was used to represent the areas which reported most sightings.

All geocoordinates were expressed in decimal degrees and imported into ArcMap 10 (ESRI, Redlands California, USA) for spatial analysis. Geographic distribution was measured as the extent of occurrence (home range size). The extent of occurrence (Lindsey *et al.* 2004) was calculated using the Minimum Convex Polygon (100% MCP) method (Worton 1987). 100% MCPs are created by joining the outermost location points and the total area within the polygon represents the individual's home range size (Worton 1995; Lindsey *et al.* 2004). Although the fixed kernel utilization distribution (UD) method is commonly preferred over the MCP technique as a home range estimator (Worton 1987; Harris *et al.* 1990; Börger *et al.* 2006), the MCP 100% method was more appropriate in this instance due to the opportunistic and probably spatially and temporally biased nature of the data collection and the small sample sizes. The Kernel UD method calculates home range size of an animal based on the relative amount of time it spends in different areas of the range (utilization distribution) (Seaman & Powell 1996). The density of points throughout its range represents the relative amount of time spent in that particular area (Seaman & Powell 1996). Therefore, density estimates will be high in areas with many location fixes and low in areas with fewer fixes (Seaman & Powell 1996). In my study, cheetah sightings were collected on an opportunistic basis, thus the area with greatest observer activity had an increased chance of cheetah sightings and was inevitably biased in terms of sighting frequency and location. In addition, the core area driven by game viewing vehicles made the likelihood of sighting cheetahs in other areas extremely low. Therefore, the density of locations would not necessarily represent the amount of time an animal spent in a particular area (Seaman & Powell 1996). Furthermore, areas with no recorded sightings do not necessarily indicate the absence of cheetah occurrence but rather that no sightings were reported (Lindsey *et al.* 2004). Absence of sightings may either mean that there were no cheetahs in that area or that they were present but not recorded.

100% MCPs (km^2) were calculated for all adults (females, including when they had cubs, single males and male coalitions) that had three or more valid location points (n = 9) (Marnewick & Davies-Mostert 2012). The locations of cubs were only recorded once they became independent from their mother and either seen alone or in a sibling group. I investigated the influence of the number of location fixes on individual/group home range sizes (100% MCP) using regression analysis (STATISTICA 12). The home range overlap between individuals/groups was determined by calculating the proportion of area shared with other cheetahs by dividing the total shared area by the home range size.

4.3 Results

A total of 447 cheetah sightings amounting to 13179 photographs were received from participants within NOTUGRE (89.0%), properties in South Africa within the Greater Mapungubwe Area (6.1%), and the Tuli Circle and Sentinel Game Farm area of Zimbabwe (4.9%). Eighty-nine sightings (19.9%) were received without photographs and 38 sightings (8.5%) did not have accompanying location data.

4.3.1 Long term trends and population demography

The frequency of sightings varied between seasons, being higher in the cool dry season (n = 264) than during the hot wet season (n = 179). Four sightings lacked accompanying dates. Thirty-two cheetahs (18 males and 14 females) of all age classes were identified within and adjacent to NOTUGRE between 2006 and 2013 (Appendix IV; Appendix V). A further 13 individuals of unknown sex were also identified, but they were only sighted once or twice and only had half profiles. Age classes at first sightings are shown in Appendix V. To maintain a conservative overall estimate, only cheetahs with left-side profiles (n = 35; Appendix IV) were included in the total estimate for NOTUGRE. To assess the general population trend, the total number of individuals identified (regardless of their frequencies of sighting) in each year was compared (Figure 4.2). Although fewer photographic and sightings records were received for the years prior to 2013 (Figure 4.2), the calculated population sizes for 2006, 2010, 2011 and 2012 were higher than 2013, although only marginally (Figure 4.2).

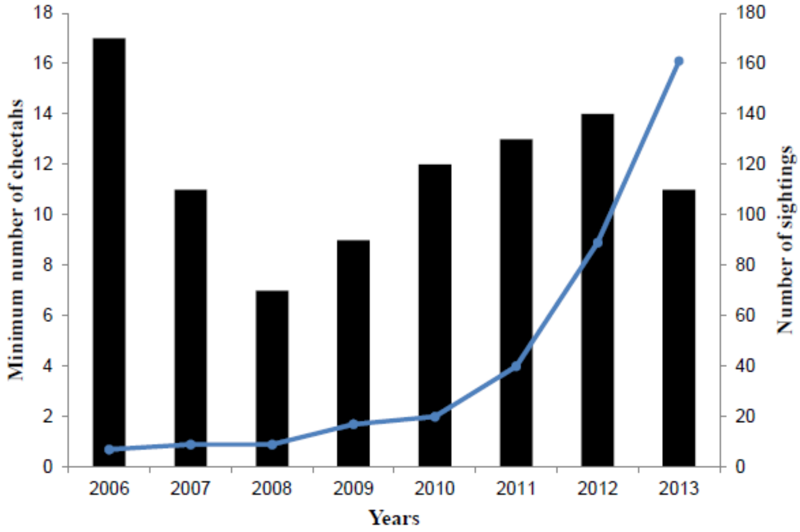


Figure 4.2 The minimum number of cheetahs identified within NOTUGRE (all age classes), Botswana between 2006 and 2013. The number of sightings per year is also indicated (blue line).

The sighting frequencies, estimated date of birth and relatedness for identified cheetahs are shown in Appendix V for individual cheetahs identified during the photographic survey period (January 2006-November 2013). The number of times individuals were sighted varied. Nineteen individuals (12 adults; 7 cubs) only had a single sighting; 26 (full and half profiles) were re-sighted at least once (> 1 sighting); four (1 adult; 3 cubs) were re-sighted twice, and 22 (10 adults; 12 cubs) were re-sighted three or more times and were therefore considered to be resident individuals (Kelly 2001). Cheetahs with two or less sightings could have been vagrants (Gros *et al.* 1996). Individuals in family groups or stable coalitions that suddenly disappeared were considered to have died (Balm *et al.* 2012).

A coalition of three males had extensive photographic records, including sightings dating back to 2006 (Appendix V). Two of the three could be confirmed as brothers (CM1 and CM2) in a litter of four as they were first photographed when they were approximately 5 – 6 months old. The same litter was photographed about eight months later with only two cubs remaining (CM1, CM2) and confirmed by two further sightings a few days later. However, the third individual (CM3) was not a littermate and joined the two litter-mates at a later stage.

The age of cheetahs was calculated at the end of 2013. CM1 and CM2 were roughly 7.5 years old. CM3 was at least 7 years old (first photographed as an adult 1 May 2008). CF1, CF2 and CF3 all appear to be long-term residents and were estimated to be at least 9, 8 and 7 years old respectively. The other cheetahs with known birth dates were all cubs and sub-adults.

Eight of the 45 (17.8%) cheetahs identified were photographed by camera trapping only, including a resident coalition of two (CM5 and CM6) with part of their territory stretching over the central part of NOTUGRE.

4.3.2 Minimum population size (NOTUGRE)

Sightings within NOTUGRE between April and October 2013 amounted to 100 sightings and 6886 photographs. Only two sightings had cheetahs that could not be identified, but another group member could clearly be identified in both of these sightings. The majority of the sightings (86%) were from the properties on which Mashatu Game Reserve operates. A total of 13 cheetahs (nine adults and four cubs) were identified; three more cubs were omitted from the estimate as they were known to have died (cause of death unknown). Thus, the minimum population size for NOTUGRE on 1 July 2013 was estimated to be 10 individuals (nine adults and one cub).

Overall, five cheetah social groups were identified that comprised nine adult cheetahs (six males and three females) and one female cub. Two male cheetahs in a coalition were detected by camera trapping only, presumably due to their skittish nature. The demographics of the NOTUGRE cheetah population were: two male coalitions (a coalition of three and a coalition of two), a sibling group (consisting of one male and one female litter-mate that had recently left their mother), one lone female, and one family group (consisting of a mother and female cub). The adult sex ratio over this period was not estimated due to the small sample size. All individuals accounted for in the minimum population size were believed to be resident as they were all sighted regularly (Appendix V) and the males were seen scent marking, a behaviour shown by resident territorial males (Caro 1994).

4.3.3 Distribution and home range

The locations of cheetah sightings were unevenly distributed across the reserve. The properties driven by the tourism operators showed a higher density of cheetah sightings (Figure 4.3). Certain properties have no commercial lodges and are rarely driven by field guides (Figure 4.3). Thus, caution needs to be exercised when interpreting the overall distribution of cheetahs, although higher densities in the reserve are expected due to lower human interference.

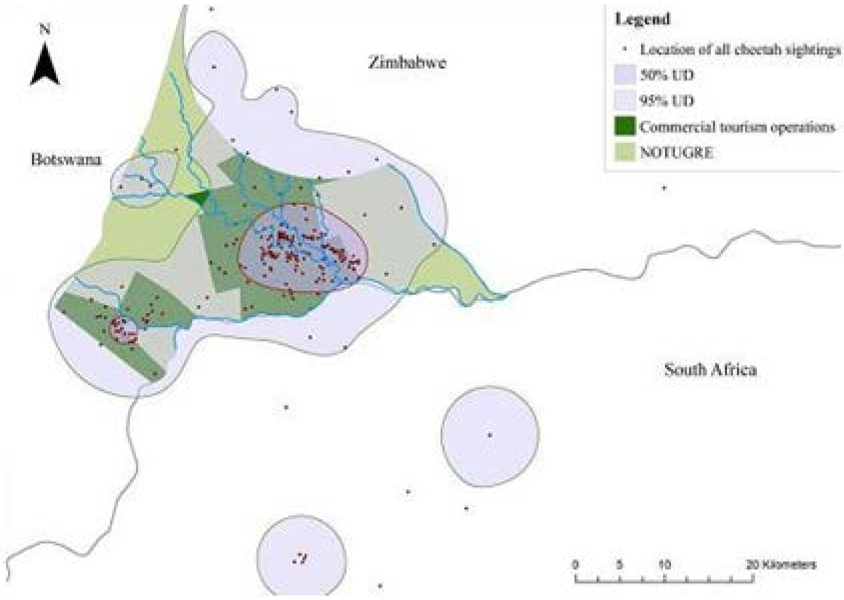


Figure 4.3 Map depicting the distribution of all cheetah sightings within and outside NOTUGRE between January 2006 and November 2013, the dark green sections are properties used by commercial tourism operations. 95% and 50% Kernel UD distribution range estimates for the location of sightings are also presented.

Location fixes were predominantly from precise GPS fixes (accuracy 1) (45%) but fixes with a precision of 2 (33%) and, or 3 (22%) were also used to increase sample sizes. Home range sizes (MCP 100%) varied markedly across their distribution (Table 4.1). The largest estimated home range was for female CF3 that used an MCP 100% of 256 km² (Figure 4.4). The second largest estimate of 188 km² was for the all-male coalition of CM1, CM2 and CM3 (Table 4.1; Figure 4.4).

Table 4.1 The extent of occurrence (MCP100%) and the number of fixes for all identifiable cheetahs with three or more location fixes. Location fixes were derived from all sightings records between January 2006 and November 2013, the dates of the first and last locations as well as timespan are also presented here. Home ranges of cubs were only calculated from time of independence.

Cheetah ID	MCP 100% (km^2)	Number of fixes	Date of first location	Date of last location	Timespan (days)
CF3	256	134	03/10/2009	15/11/2013	1505
CM1, CM2, CM3	188	72	24/01/2009	28/10/2013	1739
CF1	157	22	17/01/2006	08/01/2013	2549
CF2	155	32	18/11/2008	09/10/2012	1701
CF4	117	15	27/05/2013	22/10/2013	148
CF6	78	7	09/02/2012	12/01/2013	338
CF9	27	3	07/02/2013	30/03/2013	51
CM5 & CM6	24	7	28/06/2013	29/10/2013	124
CF12	0.21	3	06/08/2006	31/01/2007	179
Mean ±SD.	125.3±80.3	36.5±45.2			926±945.3

The centre and south of the reserve are mostly used by cheetahs with many overlapping home ranges (Figure 4.4). The western corner of the reserve was seemingly utilized only by CF6 with a little overlap with CF2, CF3 and CF4.

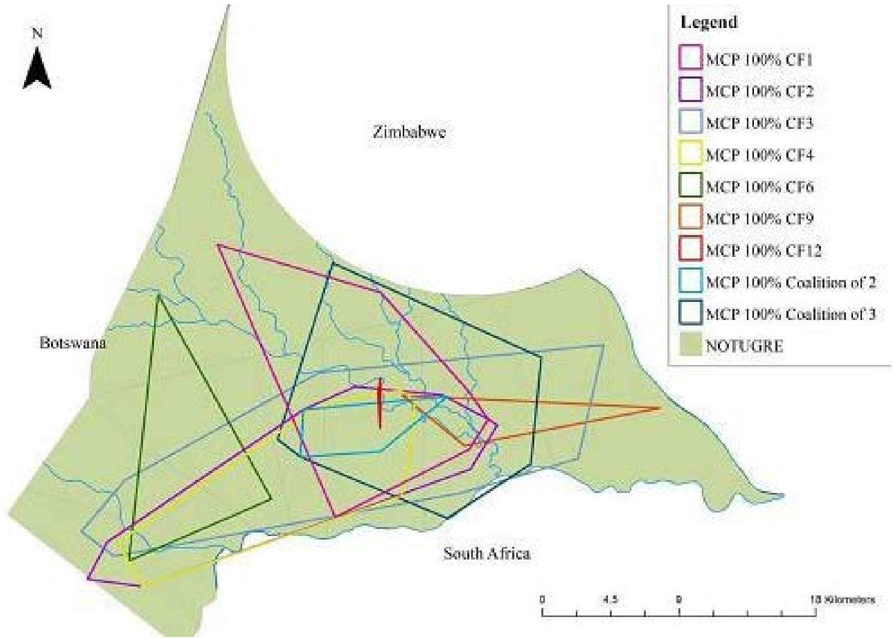


Figure 4.4 The location and extent of occurrence (MCP 100%) of all recognisable adult cheetahs and cheetah groups with three or more location points (*n* = 9) (ArcGIS 10; projected: Transverse-Mercator, spheroid WGS84, central meridian 29; mapping units: meters).

The effect of sampling bias was investigated by plotting the home range sizes (MCP 100%) against the number of location fixes (Figure 4.5). Home range sizes were significantly driven by the variation in the number of location fixes used (Figure 4.5). The greater the number of location fixes, the larger the home range size. However, the fitted curve suggests that there is an upper limit to the number of location fixes required to reach an asymptote, at approximately 120 location fixes (Figure 4.5). Nonetheless, the number of location fixes should be obtained from a wide area and home ranges calculated here must be interpreted with caution.

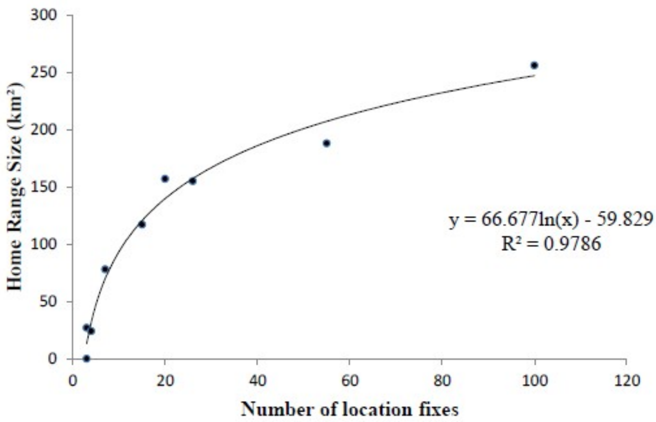


Figure 4.5 A scatterplot representing the relationship between the number of location fixes and home range size of individuals with three or more location fixes.

4.3.4 Home range overlap

The home ranges of the cheetahs overlapped considerably (Table 4.2). The two male coalitions showed wide home range overlap in both space and time (Figure 4.4); the home range of the male coalition of two (CM5; CM6) fell entirely within the home range of the male coalition of three (CM1; CM2; CM3). Independent cubs showed similar ranging patterns to their mother's. Independent female cubs CF4 and CF9 shared, respectively, 84.6% and 88.9% of their home ranges with their mother (CF3). The coalition of three males (CM1, CM2, and CM3) shared 61.2% of their home range with their mother (CF1). The home ranges of adult females overlapped between 0% and 100% (37%). Overlap of a female's home range with that of a male ranged between 0% and 100% (49%). The high home range overlap may be an artefact of the small sample sizes used to calculate some of the home ranges. Additionally, the home ranges were worked out using all available data thus includes a sampling period spanning over seven years when ranges are likely to shift.

Table 4.2 The percentage overlap for each adult cheetah's 100% MCP home range for the Northern Tuli Game Reserve for individuals alive over the same time period (January 2006 – November 2013).

Cheetah ID	Overlap							
	CF1	CF2	CF3	CF4	CF6	CF9	Coalition 2	Coalition 3
CF1	-	42%	55%	28%	0%	6%	15%	99%
CF2	43%	-	86%	75%	17%	6%	100%	44%
CF3	33%	52%	-	39%	18%	9%	9%	50%
CF4	38%	100%	85%	-	21%	0%	19%	34%
CF6	0%	35%	58%	31%	-	0%	0%	0%
CF9	37%	37%	89%	1%	0%	-	4%	67%
Coalition 2	96%	100%	100%	92%	0%	4%	-	100%

Coalition 3	61%	36%	68%	21%	0%	10%	13%	-

4.3.5 Cross boundary movement

Cross boundary movement was observed only between NOTUGRE and the Tuli Circle of Zimbabwe. Although cheetahs were sighted in South Africa (*n* = 9 individuals), there was no evidence for movement across this border. Furthermore, there were no sightings of cheetah within the Mapungubwe National Park (Cilliers, Section Ranger in Mapungubwe National Parks, pers. comm.) or the Maramani Communal lands east of NOTUGRE, in Zimbabwe. Cheetah sightings west of the reserve were also scarce (see Chapter 5).

4.4 Discussion

The photographic survey method was a useful technique to get rapid and seemingly robust population estimates for cheetahs within NOTUGRE. Public response was excellent, particularly in the properties which had the most intensive awareness. Personal communication was by far the best technique for obtaining photographic sightings and the aid of volunteers to assist with public liaison at various tourist camps should therefore be considered for future censuses. Digital cameras supplied to field guides were also an efficient tool as photographs were accompanied by GPS locations and correct dates and times. However, it is important to ensure that the guides understand the project. Providing rewards for their assistance may also facilitate their support. Tour guides, including professional photographers, are frequent visitors of game reserves and were the most useful source of historical cheetah sightings; they also showed enthusiasm and were eager to assist.

Other indirect methods such as questionnaire surveys are commonly used to provide information on cheetah populations. Questionnaires that rely solely on sightings information suffer from inaccurate reporting, including errors in aging and sexing of individuals (Gros *et al.* 1996). Furthermore, the method may underestimate population size because all sightings of the same group compositions are considered to be the same as individuals cannot be distinguished (Gros *et al.* 1996). Photographic surveys allow for individual recognition, therefore providing a much more robust method for population estimates.

Possible drawbacks to photographic surveys include uneven search effort, inaccurate reports, imprecise metadata and, despite the use of database management software, the time required to compare and match large quantities of photographs. Regardless of these disadvantages, this method provided important baseline data on the minimum population size, home ranges, ages, family relations and cross-boundary movement that would otherwise have required years of intensive research. Furthermore, once a relatively comprehensive cheetah photographic database exists (such as Appendix IV), this method can be used as a tool to monitor future cheetah population survival rates and recruitment.

Cheetah sightings were more frequent in the cool dry season. This is possibly because of a more open landscape and better visibility. In addition, the cool dry season also coincides with the peak tourism season for photographic destinations. Based on the results of this study it is recommended that future photographic surveys be carried out over the cool dry season.

4.4.1 Camera traps as an additional tool

Photographic surveys rely on photographs taken opportunistically, hence they are biased to cheetahs that are relatively habituated and which frequent the most visited areas (Gros *et al.* 1996). The method also relies on good quality photographs for identification of individuals (Gros *et al.* 1996). Most populations will have a few shy individuals that may not be documented at all (Maddock & Mills 1994; Gros *et al.* 1996). Forty-seven additional sightings with a total of 517 photographs were captured by camera traps (camera traps set out for research and recreational purposes) in my study. Camera traps photographed eight otherwise unrecorded individuals and 89% of images from South African properties were taken by camera traps. This suggests that not all individuals within a population will be detected if only the traditional photographic submissions are used. Therefore, photographic surveys should not rely solely on the use of photographs but also supplement the database with camera trapping data.

4.4.2 Minimum population size and trends

The total population sizes calculated over the last eight years (Figure 4.2) could suggest a decreasing population; however there is insufficient data to draw a conclusive interpretation. It is expected that when more effort and hence more photographic sightings are received, more individuals in the population are likely to be detected. However, despite the higher effort, including awareness, camera trapping and cameras supplied to field guides, the minimum population size for 2013 was below that of previous years. In addition, camera trapping sightings were only received for 2012 and 2013 thus shy individuals may have been excluded in the minimum count of preceding years. Possible reasons for the observed decline may be increased intra-guild predation with a recovering lion population (Snyman *et al.* 2014). A high lion density can have a negative impact on cheetah numbers, particularly if the cheetah population is fragmented (Laurenson 1994; Kelly *et al.* 1998; Kelly & Durant 2000; Durant *et al.* 2004).

Human-related mortalities and the persecution of cheetahs outside the boundaries of the reserve (O.R. Masupe, Community Liaison officer/Anti-poaching officer, pers. comm.) may also be a cause for their observed decline. During the eight year period, a number of cases of predator persecution were reported, including the death of a sub-adult cheetah reported from the community-owned farmland adjacent to NOTUGRE. Furthermore, wire snaring inside the reserve is a growing concern (P. Le Roux, general manager of Mashatu Game Reserve, pers. comm.). Although snares are mostly set to capture small to medium antelope for bushmeat, incidental capture of cheetahs have been recorded in NOTUGRE.

4.4.3 Family relations

Relationships between certain individuals could be determined from photographs of family groups. The coalition of three males is composed of two littermates and a non-relative. Caro (1994) observed similar trends and found that most coalitions with three members consisted of two littermates and a non-relative, whereas most coalitions of two are composed of littermates, with an average 29.4% of cheetah coalitions containing non-relatives (Caro 1994). More recently a genetic study conducted in Botswana found that three out of four male coalitions were genetically unrelated (Dalton *et al.* 2013).

4.4.4 Estimated age

The minimum estimated ages of the male cheetahs in my study are comparable with that of the estimated minimum ages of males in the Serengeti (6.0 to 8.4 years = 6.9, SE = 0.2) with territorial males expected to have a longer lifespan than non-resident males (Caro 1994). Adult female cheetahs in my study showed a higher lifespan (average 8 years) than that of Serengeti females (average 6.2 years; Caro 1994) but Serengeti cheetahs were also observed to reach 14 years 5 months (female) and 11 years 10 months (male) (Durant *et al.* 2010). Furthermore, the estimated ages for the adult female cheetahs in my study are probably underestimated since these cheetahs were assigned a minimum age of two when first sighted as adults.

4.4.5 Distribution and home ranges

The observed distribution of sightings identifies areas of cheetah occupancy/presence both within and outside the formally protected areas. Although the widespread distribution of sightings may include outliers such as temporal trips outside home ranges and dispersing individuals (Lindsey *et al.* 2004), my findings highlight the need for conservation action to include both protected and unprotected areas.

Most of the recorded sightings occurred on commercial properties where ecotourism is the primary land use. This highlights the effect of sampling bias as more frequent activity on these properties is evident and guides are actively searching for this species whilst on game viewing drives. Mashatu Game Reserve is a commercial photo-tourism operation, with two commercial and five private camps, and a maximum of 15 vehicles operating at any one time. Vehicles used by researchers, horse safaris, walking safaris, photographic safaris and eco-training may also operate on the reserve, and certain areas are driven more frequently than others, particularly the alluvial plains surrounding the major river systems, because of the higher abundance of game in general and carnivores in particular, therefore more photos are likely to be recorded in these areas (Figure 4.3).

Nonetheless, cheetahs have large home ranges relative to property sizes and individuals that may occur on properties seldom driven are likely to be sighted on neighbouring properties and were thus probably included in the minimum population estimate (Maddock & Mills 1994). Importantly, only a single sighting of an individual is required to include it in population estimates.

Home ranges should be interpreted with care as the MCP 100% method provides only a rudimentary, unsophisticated estimate of home range. Previously reported home ranges varied between 11 km^2 and 1651 km^2 (Pettifer 1981; Caro & Collins 1987; Mills 1989; Caro 1994; Gros *et al.* 1996; Hunter 1998; Marker 2000; Purchase & du Toit 2000; Broomhall *et al.* 2003; Bissett & Bernard 2006; Bissett 2007). Additionally, Bissett & Bernard (2006) found home range sizes to differ with different social groups and Bissett (2007) reports that home ranges of female cheetahs differ with reproductive status. In my study, the home range size for the coalition of three males (188 km^2) was considerably larger than that of the average territory for male cheetahs in the Serengeti Plains (37.4 km^2: Caro 1994). However, the largest home range size I recorded was used by a female (256 km^2) and this is smaller than the smallest female range recorded for a Serengeti cheetah (394.5 km^2: Caro 1994) but larger than home ranges from cheetahs in the KNP (102 km^2 to 192 km^2: Broomhall *et al.* 2003). Nonetheless, a very strong correlation was observed between the number of fixes and home range size (Figure 4.5). Thus, the number of fixes used may have been insufficient to accurately calculate some individuals' home ranges.

4.4.6 Home range overlap

Both male coalitions appeared to be resident and territorial as they were repeatedly seen in the same area and scent-marking at specific scent marking posts (Caro 1994). Furthermore, a scent-marking post was used by members of both coalitions. Extensive home range overlap in males has been noted in non-resident males in the Serengeti (Caro 1994), but it is not common in resident males that are territorial.

Individual cheetahs with known family relationships showed considerable spatial overlap, consistent with previous findings where sibling groups that had recently left their mother were recorded to have an average of 79.5% of their home range falling within their natal range (Caro 1994). However, the home range of CM1 and CM2 (7.5 years old) overlapped considerably with that of their mother, implying that they did not disperse from their natal home ranges. This is unusual for male cheetahs and inbreeding may potentially be a concern (Kelly *et al.* 1998). Differences in home range overlap for family and non-family members could not be assessed because of unknown relatedness between some individuals.

4.4.7 Sightings outside of NOTUGRE

Sightings outside NOTUGRE were scarce (11% of all reports) and reports were mostly from camera traps, suggesting that although cheetahs occur outside NOTUGRE, they are rare and skittish. However, restricted access to certain properties outside NOTUGRE prevented a thorough census and data were limited to reports from a few respondents (n = 6). Very few sightings in this study were reported from communally-owned farmlands that occur both east and west of the reserve (Zimbabwe and Botswana). Questionnaire surveys were undertaken in the communities west of the reserve (see Chapter 5), and few respondents (n = 24) had seen cheetahs and many farmers could not distinguish between cheetahs and leopards (*Panthera pardus*). Scarce cheetah sightings could be a result of poor observer activity or the presence of skittish individuals, or they could be attributed to depleted prey populations (see Chapter 5). However, it has been shown that human disturbance alone can limit cheetah occurrence by the alteration of natural habitat, the presence of domestic dogs (*Canis lupus familiaris*), bushmeat poaching and direct persecution (Andresen *et al.* 2014). Significantly, low prey availability is typically associated with transient carnivore populations (Winterbach *et al.* 2014). Habitat fragmentation and persecution by livestock farmers may further discourage the movement of cheetahs outside of protected areas (Lindsey *et al.* 2004).

For a population to have long-term demographic viability, it should comprise over 300 individuals (including cubs), with isolated populations at a higher risk of extinction (Durant *et al.* 2007). Most African protected areas are not large enough to contain viable populations of cheetahs and hence rely on the relocation of cheetahs outside of protected areas, either naturally or artificially (translocations) (Durant *et al.* 2007; Macdonald *et al.* 2010b). Thus, protected areas such as NOTUGRE are unlikely to support a genetically viable population of cheetahs and must therefore rely on cheetahs persisting beyond the borders of the reserve and immigrating from time to time. The conservation of cheetahs relies on their survival across extensive areas of connected habitats with heterogeneous populations of prey and predators (Durant *et al.* 2007). It is well documented that cheetahs are adversely affected by human activity and high human densities can provide an immediate barrier to the movement of cheetahs (Lindsey *et al.* 2004; Woodroffe 2000). Persecution, fences and habitat modification may limit the distribution and dispersal of cheetahs outside NOTUGRE, possibly fragmenting populations into discontinuous sub-populations. Dispersal and therefore inter-patch connectivity is essential for populations to persist (Durant *et al.* 2007; Elliot *et al.* 2014).

The results of my study suggest that there is little or no exchange with adjacent farmlands in South Africa. Furthermore, human settlements surround much of the western, southern and eastern boundaries of the reserve. Reducing direct persecution and mortality is an important focus for conservation efforts, as well as maintaining habitat connectivity and the wild prey populations. Establishing a substantial database of sightings and distribution information provides a better understanding of which areas are important for conservation and brings clarity to the conservation requirements for cheetahs in this part of Botswana.

4.5 Conclusion

Cheetah population demographics based on the recognition of individuals can provide useful, quick and relatively inexpensive population estimates. The findings of this study provide baseline data on the status of cheetahs of NOTUGRE and contribute to a better understanding of cheetah ecology for a population occurring in an open system that experiences different pressures, including human activity. Furthermore, the study demonstrates the feasibility of a photographic survey combined with camera trapping to survey a cheetah population. The use of photographic records, including recent and old photographs, may provide an alternative to intensive field studies that involve high financial and time expenses. Comparing recent and old photographs not only provides current estimates of minimum population size but can also provide information on changes in population sizes, relatedness and age of frequently seen individuals. Asking the public for photographs also raises community awareness of these animals.

Future research should focus on the monitoring of cheetahs outside of NOTUGRE to address the lack of information on the distribution and status of the population outside of the reserve, including possible connectivity to the NOTUGRE cheetahs. It is further recommended to have on-going population monitoring to understand the drivers of this population, particularly in relation to human persecution and changes in the numbers of other carnivores.

CHAPTER 5

HUMAN-PREDATOR CONFLICT IN AND AROUND THE NORTHERN TULI GAME RESERVE, BOTSWANA

5.1 Introduction

Human-wildlife conflict (HWC) is a global concern that can arise wherever humans and wildlife come into contact (Schiess-Meier *et al.* 2007). Both humans and wildlife are generally negatively affected; wildlife can threaten economic resources and human lives and the perceived or actual threat posed by wild animals often resu lts in their persecution (Marker *et al.* 2003a, 2003c; Clarke 2012). The worldwide increase in the human population is causing habitat loss and fragmentation (Ray *et al*. 2005). As human populations move into previously uninhabited areas, the potential for conflict between humans and wildlife increases as both humans and wildlife are forced to compete for the same limited resources (Graham *et al.* 2005; Clarke 2012). Importantly, conflict contributes to the decline of wildlife populations, particularly large predators, threatening the survival of species on a local, regional and, in some instances, global scale (Ogada *et al.* 2003).

The decline in wildlife populations as a result of persecution has led many countries to promulgate laws for the protection of the wildlife involved (Graham *et al.* 2005; Sifuna 2010). However, when wildlife, and predators in particular, are legally protected, farming communities often develop resentment towards authorities and wildlife conservation programs, which are often perceived as unsympathetic towards the farmers' needs (Mishra *et al.* 2003; Clarke 2012). Thus, mitigating HWC requires a thorough understanding of a complex situation and effective measures will differ from one locality to the next (Dickman 2010; Clarke 2012). Nonetheless, large carnivore populations can persist and co-exist in human-dominated landscapes where appropriate wildlife management is established and enforced (Linnell *et al.* 2001).

Human-predator conflict, particularly over the depredation of livestock, is one of the most prominent forms of HWC (Ogada *et al.* 2003; Graham *et al.* 2005). The occurrence of predators in human-inhabited areas can lead to conflict due to the perceived threat to human lives (Graham *et al.* 2005) but more frequently because of the high financial costs associated with livestock depredation (Patterson *et al.* 2004; Graham *et al.* 2005; Winterbach *et al.* 2014). Depredation of livestock may result in revenge killing of predators by farmers (Sifuna 2010; Chattha *et al.* 2013). However, persecution may also be indiscriminate and is not always retaliatory, but rather used as a means of prevention (Marker *et al.* 2003c). The killing of predators is often considered to be the most cost-effective and efficient way to reduce levels of predation on livestock (Thorn *et al.* 2014). However, Graham *et al.* (2005) found that predator density is not related to livestock depredation, but may rather be a function of prey availability, hence reducing predator abundances is unlikely to resolve the conflict. Indiscriminate persecution of predators can have a severe impact on the conservation of threatened species (Ogada *et al.* 2003; Thorn *et al.* 2014). Furthermore, ecosystems are complex, with multi-trophic interactions, therefore removing large predators from the system can trigger undesirable ecological responses (Graham *et al.* 2005). For instance, the absence of large predators on farmlands may trigger 'meso-predator release' which may further exacerbate the livestock-predator conflict (Prugh *et al.* 2009). Meso-predator release is the dramatic increase in the abundance of smaller predators which is commonly associated with the absence of large predators (Prugh *et al.* 2009).

Large predators typically occur at low densities due to their large space and energy requirements (Patterson *et al.* 2004; Ray *et al.* 2005). Protected areas may therefore be too small to support viable populations of large predators and connectivity of isolated populations across agricultural farmlands may be essential for predator survival (Ogada *et al.* 2003; Selebatso *et al.* 2008; Winterbach *et al.* 2014). Furthermore, subordinate predator species such as cheetahs (*Acinonyx jubatus*) and African wild dogs (*Lycaon pictus*) may actively seek refuge outside protected areas from more dominant competitors which generally occur at higher population densities within reserve boundaries (Laurenson *et al.* 1995; Marker 1998; Marnewick & Cilliers 2006; Klein 2007; Winterbach *et al.* 2014). For example, about half of Botswana's cheetah population is found on farmlands and African wild dogs are widespread across agricultural land (Winterbach *et al.* 2014). Human-dominated areas outside protected areas can therefore have a direct impact on the survival of these wide-ranging carnivores, and may contribute to localized extinctions (Woodroffe & Ginsberg 1998; Klein 2007; Winterbach *et al.* 2014).

The protection of large predators outside protected areas is essential and conservation action plans need to account for this (Laurenson *et al.* 1995; Woodroffe & Ginsberg 1998; Kelly & Durant 2000). Reducing predator persecution is crucial in addressing human-predator conflict (Thorn *et al.* 2012). Livestock depredation can be prevented by implementing appropriate livestock husbandry techniques and developing a better understanding of the carnivore species involved (Ogada *et al.* 2003). However, mitigating the wildlife damage, such as reducing livestock loss, alone is unlikely to resolve the conflict as social factors and attitudes towards predators are also important determinants of human behaviour (Dickman 2010). Importantly, conflict mitigation can be achieved through a change in attitude towards wildlife, encouraging cooperation and accounting for the concerns of the community, as successful conservation cannot succeed without the support of local communities that coexist with the wildlife (Sifuna 2010).

Attitude can be defined as '*a learned predisposition to respond to an object or class of objects in a consistently favourable or unfavourable way'* (Foddy 1994) and a person's attitude will directly affect their choice of action (Hudenko 2012). Thorn *et al.* (2014) found that farmers who killed predators had significantly more negative attitudes towards predators than farmers who did not kill predators. However, livestock losses to predators were not a strong determinant of farmer attitude (Thorn *et al.* 2014). Attitudes are mostly formed through evaluations of personal values and by direct knowledge, but also by emotional responses based on previous experiences and, most importantly, by social influences (i.e. discussions with neighbours) (Foddy 1994; Thorn *et al.* 2012). A positive attitude within and among farming communities may increase tolerance towards predators. Thus, a sound understanding of the elements which may form and change attitudes towards predators is essential for conservation planning (Thorn *et al.* 2014).

Socio-demographic variables such as age, gender, annual income and the level of education may impact attitudes (Røskaft *et al.* 2007). A more positive attitude towards predators is generally observed where people have a better understanding of the environment and its functioning (Røskaft *et al.* 2007). Marker *et al.* (2003a) found that Namibian farmers were less likely to persecute predators on their farms after participating in an education program about predators. But, providing information alone is unlikely to change attitudes as humans rarely make rational decisions; decision making is primarily driven by an individual's emotional response rather than by logic and reasoning (Hudenko 2012). However, generating affection and evoking positive emotions from positive experiences with wildlife can change attitudes (Hudenko 2012; Heberlein 2012). Understanding the drivers of attitude, personal values, and the emotional relationships which people have with wildlife is therefore important for any conflict mitigation strategy.

Attitudes towards wildlife can be much more positive where conservation efforts include the welfare of both the wildlife and the people (Sifuna 2010). In Botswana, the Department of Wildlife and National Parks (DWNP) is responsible for a state funded compensation scheme for livestock depredation by certain wild predator species (Schiess-Meier *et al.* 2007). This incentive was introduced in an attempt to increase tolerance towards predators and reduce HWC. Farmers are required to report losses of livestock to receive compensation; such claims are then investigated by a DWNP Problem Animal Control (PAC) officer to ensure that the damage was caused by a compensated species (Schiess-Meier *et al.* 2007). Predator species that are considered dangerous (i.e. which cannot safely be chased off) and threatened species such as the cheetahs and wild dogs are included on this list (Kgathi *et al.* 2012).

Botswana has a human population of about 2 million (annual growth rate of 1.9%) (Winterbach *et al.* 2014), of which about half live in rural villages and small homesteads (known as cattleposts) on tribal farmland. In 2008, Botswana had a livestock population of approximately 4.5 million with 92% found on rural communal farmlands (Winterbach *et al.* 2014). In 2012, the cattle (*Bos primigenius*) population was estimated to be over 3.1 million and the small stock (goats *Capra aegagrus hircus* and sheep *Ovis aries*) population was estimated to be 1.6 million (DWNP 2012). Livestock pastoralism clearly forms an important part of rural economic income and cattle have a significant cultural value by representing wealth and social standing for local people (Clarke 2012; Winterbach *et al.* 2014). However, overgrazing by cattle in rural farmlands has led to bush-encroachment, the growth of unpalatable grasses and increased proportions of bare ground, all of which will result in less available forage impacting not only the livestock grown but also the wild prey base (Myers 1975; Klein 2007). Furthermore, due to a growing human population and increased livestock numbers, farmers and their livestock are encroaching onto the edges of (and into) protected areas.

Communal farmland is situated along the western border of the Northern Tuli Game Reserve (NOTUGRE) and farmers regularly encounter large predators, including cheetahs, brown hyenas (*Hyaena brunnea*), African wild dogs, lions (*Panthera leo*), leopards (*Panthera pardus*), spotted hyenas (*Crocuta crocuta*), and black-backed jackals (*Canis mesomelas*). Depredation of livestock by these predators occurs in these communities and incidents of revenge killing and predator persecution by local farmers have been reported (O.R. Masupe, Community Liaison officer/Anti-poaching officer, pers. comm.). Human-induced mortality is also brought about by the accidental snaring of predators (Ogada *et al.* 2003). Wire snares are set both outside and inside NOTUGRE by poachers for the bushmeat trade (O.R. Masupe pers. comm.).

This chapter investigates human-predator conflict within the communal farmlands bordering onto and within NOTUGRE, and how it influences farmers' attitudes towards predators. Specifically, I assessed the extent of livestock depredation, the nature of depredation events, and how such conflict influenced attitudes towards the conservation of predators outside protected areas. I also assessed the relationships between socio-demographic characteristics (i.e. age, education level, primary source of income, position held on cattle post), livestock husbandry techniques, knowledge of local predators, livestock losses, and attitudes towards predators. I predicted that a more positive attitude would be observed among respondents who had a higher level of education and knowledge of predators, but that attitude would be negatively correlated with the extent of livestock loss (Bath 1998; Ericsson & Heberlein 2003; Røskaft *et al.* 2007). Respondents who relied on livestock as a primary form of income would be expected to have a lower tolerance of livestock loss and would therefore likely have more negative attitudes (Røskaft *et al.* 2007).

5.2 Methods

5.2.1 Study area

Communal grazing of livestock by local subsistence farmers is the predominant land use on the tribal land immediately west of NOTUGRE. The boundary between the reserve and the farmland is formed by an electrified game fence (height: 2.1m with three electrical stands at 1.8m, 50cm, 20cm) which is frequently damaged by elephants (*Loxodonta africana*) and other wildlife and therefore does not normally restrict the movements of large carnivores and/or livestock. Cattleposts (human settlements that include a few individual dwellings and livestock enclosures) are irregularly spaced across the landscape with an approximate human population density of 381 people/100km² (Klein 2007) and a cattlepost density of 21.2/100km² calculated from GPS locations acquired during the survey (Figure 5.1). A small village, Lentswe Le Moriti, is situated within the NOTUGRE boundary and the surrounding land is used for agro-pastoral farming with a total of 11 cattleposts which have livestock.

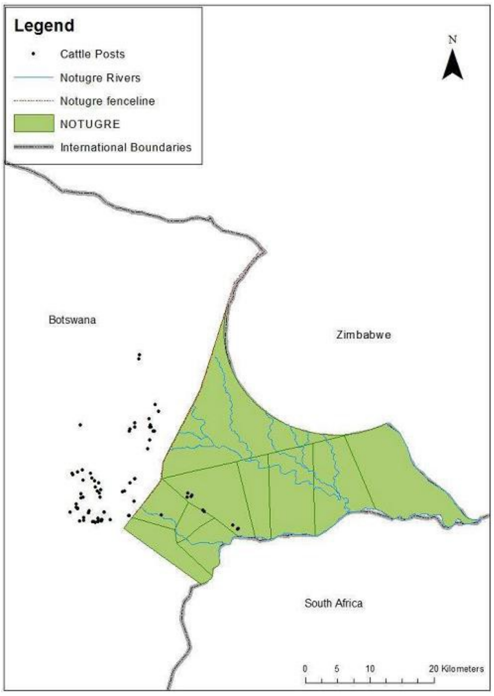


Figure 5.1 Map of the study area including the locations of all cattleposts (*n* = 80) surveyed along the western boundary of and within NOTUGRE (ArcGIS 10; projected: Transverse- Mercator, spheroid WGS84, central meridian 29; map units: meters).

The habitat in the communal farmland is generally much more open (in the horizontal plane) than in NOTUGRE as a result of over-grazing, tree felling and bush clearing (Figure 5.2; Figure 5.3). Although wildlife is present on the communal farmlands, numbers are generally low due to habitat degradation and poaching. The main occupation of residents living on cattleposts is subsistence farming and livestock pastoralism. Livestock kept includes goats, sheep and cattle. Donkeys (*Equus africanus asinus*) and horses (*Equus ferus caballus*) are also kept for transport purposes and poultry (*Gallus gallus domesticus*) is important for home consumption. Livestock is mostly left unprotected during the day, but is brought back into kraals (the traditional name for livestock enclosures, pens or corrals made up of wooden posts and/or branches) at night. However, stray animals often sleep out in the field unprotected.

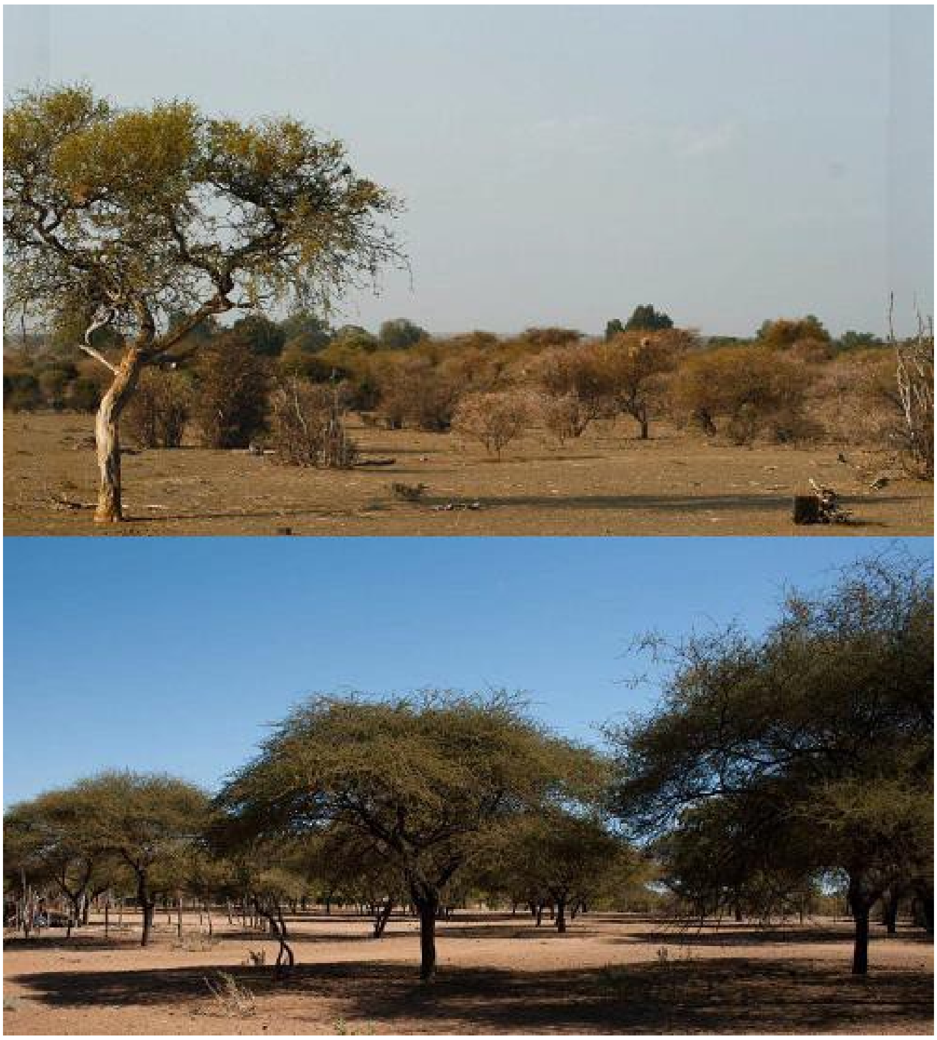


Figure 5.2 Photographs of typical acacia veld inside (top) and outside (bottom) NOTUGRE. The overall habitat inside the reserve is clearly denser with few tall trees; outside the reserve it is typically more open with little undergrowth and a distinct browse line. Photos: Eleanor Brassine.

Figure 5.3 Mopane bushveld is typically denser and has stunted growth inside the reserve (top), whereas outside the reserve the habitat is typically more open with taller trees (bottom) as a result of bush clearing and overgrazing. The absence or infrequent occurrence of elephants on farmlands also contributes to taller mopane trees which are mostly absent in the reserve. Photos: Eleanor Brassine.

5.2.2 Data collection

Information to understand the extent and potential drivers of human-predator conflict in the communities along the western boundary of, and within, NOTUGRE was collected by means of an interview-based questionnaire (Appendix VI). The questionnaire consisted of 88 questions and was divided into five sections: demographic and socio-economic characteristics (n = 8 questions); farm details and management practices (n = 36 questions); details of wildlife and predators in the area (n = 14 questions); predation and conflict (n = 25 questions); perceptions and attitudes towards predators (*n* = 5 questions). The questionnaire was written in English but interviews were conducted by translators in the local language, Setswana, where necessary. Two teams conducted the interviews simultaneously, with each team consisting of a trained researcher and a local Motswana translator who had a good understanding of local traditions and farming practices. The questionnaire survey was carried out with the authorisation of the Rhodes University Ethical Standards Committee (Ethics clearance number: ZOOL-03-2012). The questionnaire was explained and read over with the assistants prior to the start of the survey. No pilot study was conducted due to the limited number of cattleposts (*n* = 80) in the area, but the questionnaire was adapted from a similar study done in the west of Botswana by Cheetah Conservation Botswana (CCB) (Klein 2013). Furthermore, the questionnaire was reviewed by experts in the field from Rhodes University prior to being conducted (Olson 2010).

The study was conducted in the cool dry season between May and August 2012. To ensure even coverage, and to have as large a sample size as possible, all cattleposts situated within 14 km of the border of NOTUGRE were surveyed opportunistically (Figure 5.1). A previous study found that HWC occurred most frequently in areas immediately adjacent to protected areas, with the highest incidents of damage within five kilometres and up to about 20 kilometres from wildlife areas (Sifuna 2010). Cattleposts were located by asking for directions from local residents. The interviews were conducted at cattleposts within the following farming areas: Lentswe Le Moriti, Fairfields, Mathlabaneng, Sethoba, Malopeng, Letswerang, Motswereng, Monyemotobo, Lekono, Makadibeng, Semphane, Thune, Matshekge, Madiope, Thebele, Manyehome, and Mokalati. When occupants of a cattlepost were absent, we would return to it on the following visit (approximate time between visits was 1 week).

All respondents were interviewed at their cattleposts and the date, time and location (GPS, Garmin GPSMAP 62) were recorded for each interview. Visiting the cattleposts themselves allowed for an improved understanding of current farming practices and the methods used to protect livestock from predators, including the designs of kraals and the distance of kraals from homesteads. Once the interview was complete, we also took the opportunity to advise on livestock loss prevention measures, how to identify the more common predators and the status of cheetahs and their conservation.

Upon arrival at a cattlepost, we would introduce ourselves and explain the nature of the research. Residents were asked if they wanted to participate in the study, explaining that they had the right to refuse being interviewed. All of the cattlepost residents we visited (*n* = 80) agreed to take part in the questionnaire survey, with each interview lasting an average of 45 minutes. Before commencement of the interview, two posters depicting the common large predators were presented as supplementary material (Appendix VII) to ensure respondents could identify common predator species correctly and to assess knowledge of the common predators (Gros 2002). The posters consisted of photographs of eight common large predators, lion, leopard, cheetah, spotted hyena, brown hyena, African wild dog, blackbacked jackal and domestic dog (*Canis lupus familiaris*) (see Appendix VII). The questionnaire had both closed and open-ended questions, respondents could also provide additional comments if they wished. All answers to open-ended questions and any comments were recorded in full but later classified into groups according to their similarities to facilitate statistical analyses (Foddy 1994). Classification may be subject to a degree of interpretation, but a standard approach was applied when classifying responses by the principal researcher. This involved reading over and sorting all the responses into relevant categories (Foddy 1994). The respondents were made aware that they could respond with “I don’t know” to any question (White *et al.* 2005). Furthermore, to assess the accuracy of responses, ground-truthing questions (n = 15) were included which cross-referenced respondent answers (White *et al.* 2005). For example, respondents were asked to explain how their livestock was cared for during the day and night (Appendix VI, section D: questions 17 – 18). Later in the questionnaire (Appendix VI, section H; questions 63- 72), respondents were asked to provide a ‘yes’ or ‘no’ answer for the livestock husbandry techniques which they were using to protect their livestock from predators. If their answers differed, we would ask respondents to clarify their previous answer, and in the instances where the responses remained inconsistent, their questionnaire was disregarded and removed from all analyses.

Respondents were asked to give details on the number and type of livestock owned, they were then asked if they suffered any losses to predators, and if so, to give a detailed explanation on each depredation incident over the previous 18 months. A period of 18 months was chosen as it coincided to the beginning of 2011 and was thus easier to explain to respondents. Each respondent was asked to provide the estimated value of livestock in Botswana Pula (BWP), and the average value for each livestock type was calculated from all responses to calculate the total value of livestock lost per cattlepost.

Respondents were asked to rank the significance of potential problems faced as a livestock farmer on a three-point Likert scale (0 – 3). A maximum of three was given for major problems and zero was given if it was not a problem at all, thus more than one cause could be ranked as a major cause of livestock loss. The importance of each possible problem was evaluated by summing the number of maximum values for each cause. We then asked respondents to rank problem predators and to name any other predator species that they identified as damage causing species.

We asked respondents to provide details on wildlife and predator species occurring in the area and whether they perceived any changes in abundances over the last 10 years. Specific details on cheetah sightings were also requested, including the number of animals seen, the location and date. Sightings data were used to assess the occurrence of cheetahs outside the reserve and, depending on the quality of the responses, to crudely estimate abundance (see Chapter 4). Cheetah sighting details were only used if respondents could clearly identify cheetahs from the poster and could provide additional description on the behaviour of cheetahs.

The location of each cattlepost was mapped (Figure 5.1) and distances (km) to the nearest fence line boundary were calculated using ArcMap 10.0 (ESRI, Redlands, California, USA). Cattleposts found within the reserve were assigned a distance of 0 km.

5.2.3 Conflict of responses

Although the purpose of the research was clearly stated upon arriving at the cattleposts, I wore the Mashatu Game Reserve uniform and drove a Mashatu vehicle with Mashatu Game Reserve and NOTUGRE research stickers clearly visible. Respondent attitude and the answers given may have therefore been influenced by the fact that I was clearly associated with the reserve. While some respondents were unhappy with the reserve, others were grateful to see that the reserve was concerned about their problems with livestock depredation. In an attempt to counteract these problems, I would retain a neutral position, remaining objective and detached to encourage respondents to provide an honest response to my questions when conducting interviews. However, some of the answers (e.g. Question 60: Classify the predators according to the level of problem) may have been exaggerated as respondents were made aware that my research focus was primarily on cheetahs. Furthermore, respondents may have supplied answers when they did not actually understand the question or had little knowledge on the topic, such as the occurrence of wildlife abundances. The way the questionnaire was structured, with the use of ground-truthing questions, reduces this bias (White *et al.* 2005) but nonetheless care must be taken when reviewing these answers.

5.2.4 Data analysis

Descriptive statistics (means, standard deviations, frequencies and ranges) were used to explain the various results. Furthermore, the responses to certain questions/statements were used to calculate index scores by allocating values of +1, 0 and -1 to the different statements depending on the responses (Foddy 1994; Walpole & Goodwin 2001). An attitude score was calculated for each respondent and this represented their overall attitude towards predators (Walpole &

Goodwin 2001; Parker *et al.* 2014; Thorn *et al.* 2014). The attitude index was based on the sum of the scores of six relevant questions/statements. Positive responses received a +1, negative responses received -1 and ambiguous or uncertain answers received a 0 (Walpole & Goodwin 2001). For example, respondents were asked "Do you think wildlife is a natural resource to be protected?" If they answered yes, they received +1, if they answered no, they received -1 and if they were uncertain, they received zero. Thus, respondents with higher scores generally had a more positive overall attitude towards predators. A possible maximum value of +6 and a minimum value of -6 could be attained.

The same approach was adopted to generate a husbandry and a knowledge index. The husbandry index was calculated based on the answers given to 29 questions/statements that were designed to assess the suitability of livestock husbandry methods which were employed by respondents to protect and manage their livestock. Non-lethal methods of protecting livestock accrued a positive score (i.e. using herders or guard dogs, fetching livestock from the field, burning fires around the kraal), whereas lethal methods (i.e. poisoning carcasses or hunting predators) would accrue a negative score. The index scoring system also took into account the actions taken by respondents when livestock was lost to a predator, the use of a calving season and maternity kraals, accurate record keeping, and the kraaling of livestock at night. In the instances where farmers did not have cattle or small stock (goats and sheep) the response was left blank and not included in the calculations. The husbandry index was the sum of points obtained for each question/statement and could accrue a maximum value of +29 and a minimum value of -29. A high index score would indicate a better approach at protecting livestock and good farm management practices.

The knowledge index was calculated based on the amount of local knowledge of wild predators, including the current legislation regarding the protection of cheetahs and other wildlife. A high index score reflected a better understanding of the role and importance of predators in ecosystems. Furthermore, two posters depicting eight common large predators were used and respondents were asked to correctly identify each predator (Appendix VII). A total of 14 questions/statements were used to generate the knowledge index score, with a possible maximum value of +14 and a minimum value of -14.

5.2.5 AIC analyses - attitude index

To assess the relative contribution of different predictor variables on the three indices, I employed a model building approach using Akaike's Information Criterion (AIC; Akaike 1974; Burnham & Anderson 2002). The second-order Akaike's Information Criterion (AICc) was used for the dataset to accommodate for the small sample size (Burnham & Anderson 2002). Eight predictor variables were used to assess their potential effect on respondent attitude. The predictor variables tested included three categorical variables; education level (four levels), primary source of income (three levels), and position held on the cattlepost (three levels), and five continuous variables; age, the distance of the cattlepost to the NOTUGRE fence line (km), the total number of livestock lost, knowledge and husbandry indices. Two respondents were removed from the dataset prior to the analysis because three or more questions had not been answered. A total of 78 valid respondents were thus used in the overall multi-model analysis to identify the relative importance of the eight predictor variables on attitude.

Husbandry index

Nine predictor variables were used to assess their potential impact on the husbandry index. I included the following socio-demographic predictor variables: age, the number of years respondents had lived in the area, the level of education (categorical variable; four levels) and the primary form of income (categorical variable; three levels). The total number of livestock lost and the distance (km) of the cattlepost from the reserve fence-line, characteristics of kraal design - materials used (categorical variable; five levels) and maximum gap size (cm), and common circumstances of depredation (categorical: two levels - inside or outside the kraal) were also included as predictor variables. Fifteen respondents were removed from the dataset as one or more values were missing, leaving a total of 65 respondents for the analysis.

Knowledge index

Six predictor variables were used to predict the knowledge index of respondents. Five continuous variables (husbandry index, distance from the reserve fence-line (km), age, number of years respondents had lived in the area, and total number of livestock lost to predators) and one categorical variable, education (four levels), were used. Three respondents were removed from the dataset due to missing values leaving a total of 77 respondents for the analysis. Prior to the model building, Shapiro-Wilk Normality Tests were conducted to test for normality, and generalised Variance Inflation Factors (VIF) were used to detect possible multi-colinearity for all predictor variables (Freckleton 2011). Variables that had a VIF of > 5 were removed to resolve any co-linearity (Freckleton 2011). I conducted Generalized Linear Models (GLM) to test for the best combination of predictor variables for the indices. Delta AICc (ΔAICc) values and Akaike weights (*wi*) were calculated for each model and were used to explain the strength of each model relative to the other models and to assess the importance of the individual predictor variables (Burnham & Anderson 2002). The best model is expressed as the model with the lowest ΔAICc value. However, any model with a low (< 2) ΔAICc value indicates that it may be suitable (Burnham & Anderson 2002). Akaike weights also provide a measure of strength of evidence, with higher values indicating the better model suitability (Rowe 2009). Thus, all models with ΔAICc values < 2 were used, the predictor variables that featured in these models were then selected and used in a cross validation GLM to identify the best predictor variable/combination of predictor variables. All analyses were conducted using the statistical software program R version 3.0.2 and RStudio version 0.98.501 with the packages “car” and “MuMIn” (R Core Team 2013).

5.3 Results

5.3.1 Demographics of respondents:

A total of 80 respondents on cattleposts were interviewed which, to the best of my knowledge, represented all of the cattleposts within a 14 km buffer of NOTUGRE. Cattleposts that were abandoned or that did not own livestock were not included or interviewed. One of the questionnaires had to be removed as the respondent was inconsistent in his answers, thus 79 valid questionnaires were used for the analyses. All respondents were Motswana nationals and of black African descent. Seventy-two percent of respondents were female and 27.6% male. On average, each cattlepost housed 4.6 ± 3.5 persons (range: 1-15). More than 40% of respondents were in the age group of over 50 years. Less than 8% were younger than 21, with the average age being 47 ±16.5 years old (range: 14-77 years old) (Figure 5.4). A third of the respondents (35.4%), had no form of education, and there was a steady decrease in the number of respondents who had any higher levels of education. Only 5.1 % of respondents had some form of tertiary education (Figure 5.5).

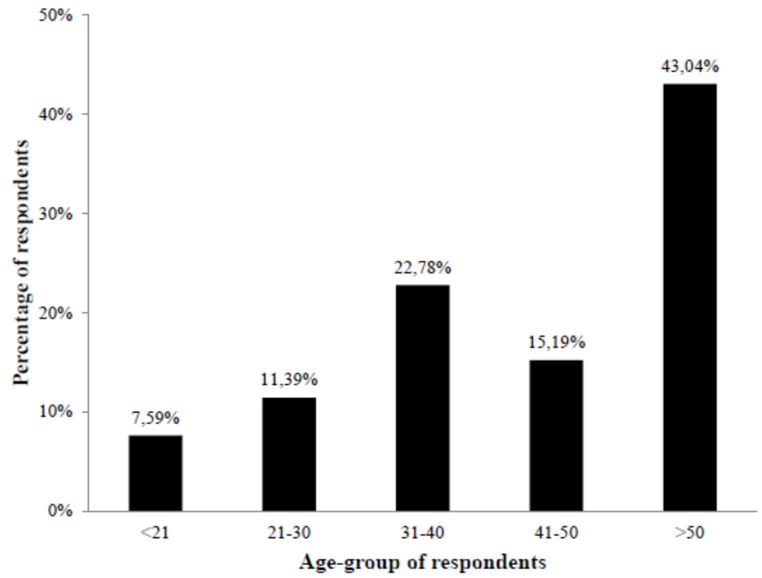


Figure 5.4 The proportion of respondents (n = 79) in each of the five age group categories.

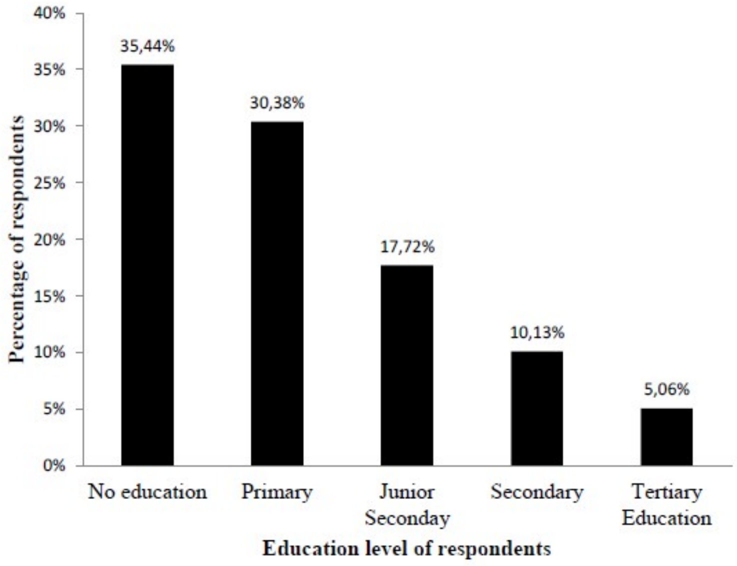


Figure 5.5 The highest education level attained by respondents (*n* = 79) living on cattleposts in communal farmlands west of NOTUGRE, expressed as percentages.

Livestock pastoralism was the primary source of income for the majority (62.7%) of respondents (Figure 5.6). Some respondents were employed by NOTUGRE and small proportions relied on other sources of income such as basket weaving, crop farming, palm beer brewing and government pensions (Figure 5.6).

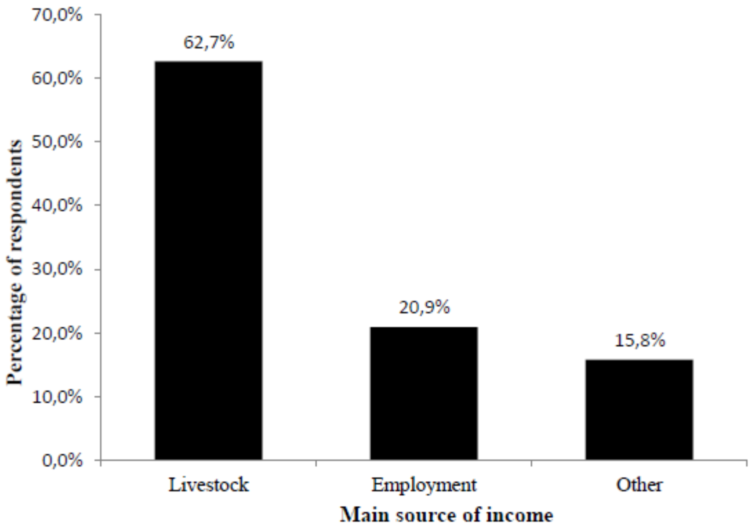


Figure 5.6 Main sources of family income for respondents living on cattleposts along the western boundary of and within NOTUGRE.

5.3.2 Livestock husbandry

A total of 6987 individual heads of livestock were owned by respondent's households. Each cattlepost had, on average, 53.13 ± 66.64 goats (range: 5-361), 21.26 ± 42.14 cattle (range: 0-280), 10.84 ± 16.95 sheep (range: 0-90), 3.1 ± 3.63 donkeys (range: 0-17), and 0.08 ± 0.58 horses (range: 0-5). The mean number of domestic dogs owned by each respondent was 2.99 ± 2.55 (range: 0-16), and chickens (10.74 ± 10.05/respondent; range: 0-50) also formed an important part of the agricultural livelihoods on the cattleposts. Grazing land outside of the reserve is typically communal and mostly unfenced. Livestock were typically not fetched by a herder from grazing in the afternoon but left to return on their own. Only 3.8% of farmers had cattle in fenced fields. During the day cattle were almost always left unattended (92.5%) with only 3.8% of respondents using herders to accompany the cattle (Table 5.1). Kraaling of cattle at night was used by 64.2% of respondents, with about a third (35.9%) leaving their cattle out of the kraal and unprotected.

Only one respondent kept his herd of small stock in a fenced field (Table 5.1). Eighteen percent of respondents used guard dogs to protect their small stock during the day, while the majority (81.3%) of farmers left their small stock unattended. Kraaling of small stock at night was used by all respondents and 20.0% of respondents placed livestock guard dogs (LGD) together with the small stock in the kraals. Most LGDs were local Tswana mixed breeds of medium size (11-25kg) and all respondents that used dogs perceived them to be effective at protecting the livestock from depredation.

Table 5.1 Summary of livestock management practices for cattle and small stock during the day and at night in communal farmlands west of and within NOTUGRE shown as a percentage of the number of respondents who owned livestock (cattle n = 53; small stock n = 79).

Day	**Cattle**	**Small stock**
Fenced field/kraal	3.8%	1.3%
Herder and guard dog	0.0%	1.3%
Herder	3.8%	0.0%
Guard dog	0.0%	16.3%
Free roaming (unattended)	92.5%	81.3%
Night		
Kraal and guard dog	1.9%	20.0%
Kraal	62.3%	80.0%
Free roaming (unattended)	35.9%	0.0%

5.3.3 Kraal design

Cattle kraals consisted mostly of horizontal poles (split-rail fence) with large gaps (38.1%) or were fenced enclosures (33.3%; Figure 5.7). Other kraal designs included vertical wooden posts (14.3%), acacia branches (7.1%), or a combination of posts and fencing (7.1%; Figure 5.7). Illustrations of the different kraal designs are shown in Appendix VIII. The average distance of cattle kraals from the homestead was 54.58m ± 54.83 (range: 0-1000m), the average height of kraals was 1.52 m ± 0.43 (range: 0.3-2.5m) and the maximum gap size between individual poles was 38.95cm ± 31.04 (range: 0-100cm).

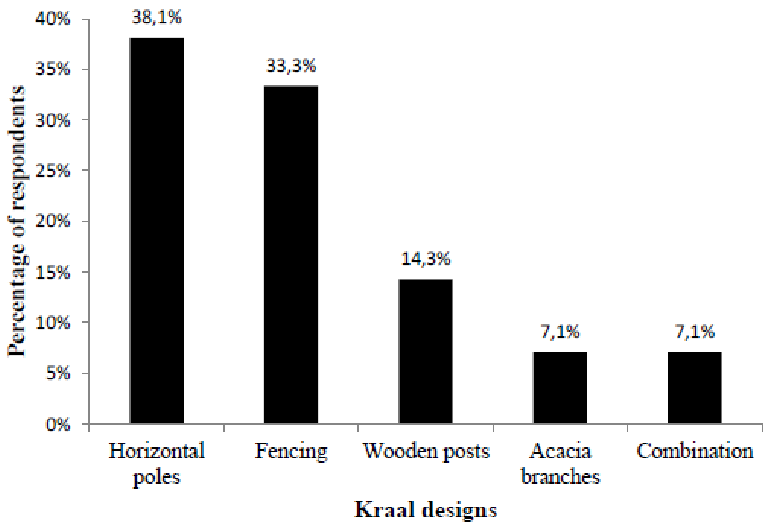


Figure 5.7 The different cattle kraal designs and materials utilised by livestock farmers for protecting their cattle at night. Percentages were calculated based on the total number of respondents who owned cattle (n = 53).

Small stock kraals were mostly (52.7%) made using vertical wooden posts (Figure 5.8). Fencing (13.5%), fencing with diamond mesh (9.5%), acacia branches (4.1%), horizontal poles (4.1%) or a combination of any of these (16.2%) (Figure 5.8; Appendix VIII). The average distance between small stock kraals and homesteads was 80.75m ± 186.46 (range 0- 1000m). Kraals were built at an average height of 1.43m ± 0.46 (range: 0.5-2.5m) with maximum gaps of 11.92cm ± 9.15 (range: 0-35cm).

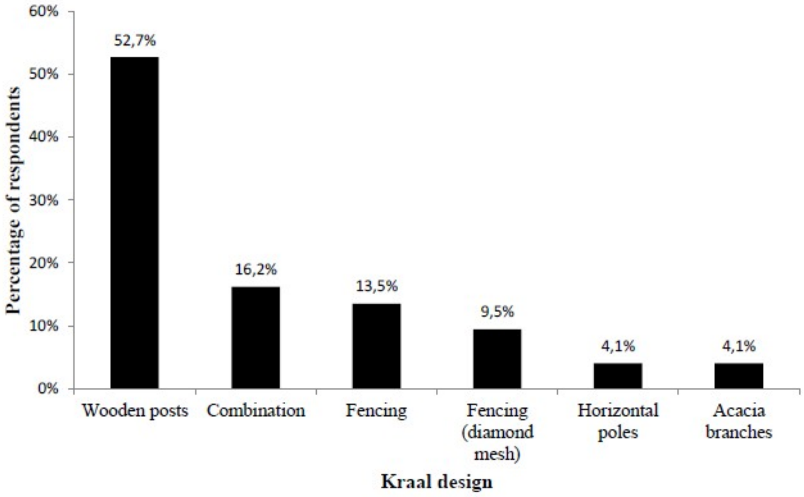


Figure 5.8 The proportions of six categories of kraal designs and the material utilized by livestock farmers (n = 79) for protecting small livestock at night.

5.3.4 Wildlife and predators

Respondents were asked to give their opinion on the status and trends of wildlife (specifically predator) species that exist in the area. Baboons (*Papio ursinus*) and honey badgers (*Mellivora capensis*) were included as predators as they were frequently mentioned as important predators of livestock and poultry. Overall, knowledge of the occurrences of wildlife species was poor with most respondents unsure of their occurrence and/or general trends over the last ten years. Eland (*Tragelaphus oryx*), zebra (*Equus burchellii*), and wildebeest (*Connochaetes taurinus*) were considered absent by more than 80% of respondents. Steenbok (*Raphicerus campestris*), scrub hare (*Lepus saxatilis*) and Helmeted Guineafowl (*Numida meleagris*) were described as being common by > 43% of respondents. However, very few respondents had an opinion on the trends in wildlife populations (64.9%) and of those that gave an answer, most indicated that wildlife populations were stable (13.5%) or increasing (12.8%), although increasing populations were often based on the presence of juvenile animals and not on the average number seen over time. These figures included respondents that had cattleposts within the game reserve boundary (*n* = 11) and where wildlife is likely to be more abundant. In order to have a better understanding of wildlife occurrence outside of the reserve, I excluded the responses for cattleposts that occurred within the reserve. Table 5.2 provides a summary of the responses for wildlife occurrence and trends excluding cattleposts that were within the reserve boundary.

Table 5.2 Summary of reported local abundance of wildlife species and population trends over the last 10 years (n = 69). Wildlife details and trends exclude cattleposts within the game reserve fence (n = 11).

Status	Kudu	Impala	Eland	Zebra	Wilde-beest	Duiker	Steenbok	Warthog	Hare	Guinea-fowl
Absent	46.8%	25.3%	79.7%	78.5%	81.0%	10.1%	17.7%	49.4%	2.5%	11.4%
Rare	20.3%	16.5%	3.8%	2.5%	2.5%	16.5%	15.2%	17.7%	3.8%	20.3%
Common	12.7%	32.9%	0.0%	2.5%	0.0%	40.5%	39.2%	15.2%	36.7%	38.0%
Very common	3.8%	8.9%	0.0%	0.0%	0.0%	16.5%	11.4%	1.3%	40.5%	13.9%
Don't know	2.5%	2.5%	2.5%	2.5%	2.5%	2.5%	2.5%	2.5%	2.5%	2.5%
Trends										
Increasing	11.6%	17.4%	2.9%	4.3%	2.9%	17.4%	15.9%	8.7%	24.6%	21.7%
Decreasing	14.5%	17.4%	1.4%	5.8%	7.2%	8.7%	8.7%	7.2%	10.1%	7.2%
Stable	8.7%	15.9%	14.5%	14.5%	15.9%	18.8%	11.6%	11.6%	10.1%	13.0%
Don't know	65.2%	49.3%	81.2%	75.4%	73.9%	55.1%	63.8%	72.5%	55.1%	58.0%

Large predators were believed to be mostly absent on communal farmland apart from spotted hyenas, baboons and black-backed jackals which were seen on an almost a daily basis (Table 5.3). The majority of respondents (73.0%) were unsure of predator population trends, except for spotted hyenas and black-backed jackals which were believed to be increasing (Table 5.3). The presence and trends of predators were mostly from visual observations (49.9%), followed by tracks (30.6%) and then calls (19.5%).

Table 5.3 Summary of the frequency of sightings of the common predator species seen by respondents. Perceived predator trends over the last ten years are also included, given as a percentage of all responses. Answers from all cattleposts were included (*n* = 79).

Frequency of sightings	**PREDATORS**								
	Lion	Cheetah	Leopard	Spotted hyena	Brown hyena	Wild dog	Baboon	Black-backed Jackal	Honey badger
Never	47.4%	71.8%	48.7%	3.9%	50.7%	92.2%	21.8%	11.7%	25.0%
< Once/year	10.3%	9.0%	3.8%	0.0%	4.1%	2.6%	0.0%	0.0%	10.0%
Once/year	12.8%	6.4%	3.8%	0.0%	5.5%	0.0%	3.8%	0.0%	8.3%
A few times a year	7.7%	3.8%	9.0%	1.3%	1.4%	2.6%	2.6%	0.0%	8.3%
Every few months	11.5%	5.1%	3.8%	1.3%	4.1%	2.6%	1.3%	1.3%	10.0%
Once/month	7.7%	1.3%	14.1%	1.3%	2.7%	0.0%	10.3%	9.1%	10.0%
Every week	1.3%	2.6%	6.4%	19.7%	5.5%	0.0%	11.5%	15.6%	18.3%
Everyday	1.3%	0.0%	10.3%	72.4%	26.0%	0.0%	48.7%	62.3%	10.0%
Trends									
Decreasing	15.2%	6.3%	8.9%	1.3%	1.3%	6.3%	5.1%	5.1%	1.3%

5.3.5 Livestock predation

Respondents were asked to provide information on incidents of predator attacks on livestock over the preceding 18 months (Table 5.4). Sixty seven out of 79 (84.8%) respondents claimed to have lost a total of 685 livestock to predators in 704 separate incidents. Furthermore, predators were also involved in the depredation of chickens and domestic dogs.

Table 5.4 The composition of livestock owned over the survey period and livestock losses to predators over the preceding 18 months (Jan 2011 – June 2012). A conservative approach was used whereby if an unknown number of livestock was lost the incident was excluded from the total count. Injured livestock were included in the counts as they often did not survive.

	Livestock					
	Goat	Cattle	Sheep	Donkey	Horse	**Total**
Livestock owned	4197	1701	835	248	6	**6987**
Total livestock lost	556	63	31	52	0	**702**
	(11.7%)	(3.6%)	(3.6%)	(17.3%)	(0%)	**(9. 1%)**
Losses attributed to predators						
Hyena*	154	41 1	6	45	0	**238**
Leopard	37	7	6	0	0	**50**
Lion	10	14	0	7	0	**31**
Cheetah	30	1	0	0	0	**31**
Wild dog	4	0	0	0	0	**4**
Baboon	101	0	0	0	0	**101**
Black-backed jackal	141	0	9	0	0	**150**
Caracal	18	0	0	0	0	**18**
Honey badger	2	0	0	0	0	**2**
Bird of prey	130	0	0	0	0	**13**
Unknown predator	46	0	0	0	0	**46**

*the name hyena is used for both species (spotted hyena and brown hyena) as respondents often did not specify although it is likely that they were referring to spotted hyenas which are far more common.

Additionally, a goat was killed by an elephant (but this was not considered as predation), five domestic dogs were killed by brown hyenas and one domestic dog was killed by a leopard. The mean number of livestock lost per cattlepost was 8.78 ± 12.57 (range: 0-92) for the 18 month period, with a monthly average of 0.49 livestock lost per cattlepost. The number of livestock lost represented approximately 9.1% (range: 0-79%) of total livestock owned (total number of livestock owned = current number of livestock + livestock depredated). Sixteen respondents suffered losses exceeding 25% of the total number of livestock owned. The highest percentage of losses suffered was 79.2% (19 animals).

Hyenas were most frequently blamed for depredation events with a total of 238 livestock believed to have been killed (33.9%), followed by black-backed jackal in 150 incidents (21.4%), baboons in 101 incidents (14.4%), and 51 leopard incidents (7.1%). Cheetahs were reported to have been responsible for 31 incidents (4.4%). More specifically, results from incident analysis showed that hyenas allegedly contributed to 65.1% of all cattle killed, 86.5% of all donkeys and 28.7% of all small stock depredation incidents. Although hyenas were believed to be responsible for the majority of attacks on all livestock types, black-backed jackals were also blamed for a number of attacks of small stock (25.6%) and baboons contributed to 17.2% of small stock attacks. Black-backed jackals are capable of attacking smaller livestock, particularly lambs and kids, left unattended and large male baboons were mostly responsible for killing lambs and kids in kraals that were left unprotected during the day (Respondents, pers. comm.). Other carnivore species that were involved in incidents of livestock depredation (including poultry) included lions, caracals (*Caracal caracal*), birds of prey (*Accipitridae* spp.), honey badgers, African wild dogs, civets (*Civettictis civetta*), mongooses (*Herpestidae* spp.), and African wild cats (*Felis silvestris lybica*). However, predator culpability may have been questionable as the methods used to identify the predator responsible for the predation event were mostly through the evidence of tracks (45.5%) or visual sight of the species (27.5%). Killing bites and feeding style, which are the more acceptable techniques, were rarely used as means of identification (15.7%). Interestingly, 8.4% of incidents were blamed on predators with no supporting evidence as the livestock was simply missing.

Perceived predator threat reflects the results of incidents of livestock depredation. Respondents were asked to rank predators according to most problem causing species. Hyenas were identified as the most problematic predator by the majority of respondents (54%), black-backed jackals were classified as the highest ranking by 20.0% of respondents followed by baboons at 15.0%.

Livestock loss was evaluated in terms of economic loss based on the following livestock values. Cattle were valued at 3500 BWP (Botswana Pula) (367 US Dollar USD) by respondents, goats at 500 BWP (53 USD) sheep at 700 BWP (73 USD) and donkeys at 400 BWP (42 USD). The total value of the livestock lost in the preceding 18 months was therefore valued at 540 500 BWP (56 699 USD). However, reliable record keeping on livestock losses was poor and most (95.5%) of the records were from memory. Smallstock were most frequently depredated and accounted for 55.4% of total economic loss. The total cattle loss was valued at 23130 USD representing 40.8%, yet the number of cattle depredated only made up 9.0% of total livestock loss. Table 5.5 tallies the economic value for livestock depredation by each predator species and gives a representation of the contribution by each predator. In terms of economic value hyenas were responsible for 46.1% of the total economic losses to all predators, which is estimated at 26 141 USD. Black-backed jackals were found to be responsible for 14.2% of total economic costs and, interestingly, lions were the third most important predator accounting for 10.5 % of the total economic losses to all predators.

Table 5.5 The total economic costs and contribution (%) of total livestock depredation, by each predator species and livestock type. Values are presented in US$ at 1US$ = 9.53 BWP (Mid-market rates: 2014-11-16 08:29 UTC).

	Livestock						
	Goat	Cattle	Sheep	Donkey	Horse	**Total**	**%**
Hyena	8025	15053	1175	1888	0	**26141**	**46.1**
Leopard	1941	2570	441	0	0	**4951**	**8.7**
Lion	525	5140	0	294	0	**5958**	**10.5**
Cheetah	1574	367	0	0	0	**1941**	**3.4**
Wild dog	210	0	0	0	0	**210**	**0.4**
Baboon	5297	0	0	0	0	**5297**	**9.3**
Black-backed jackal	7395	0	661	0	0	**8056**	**14.2**
Caracal	944	0	0	0	0	**944**	**1.7**
Honey badger	105	0	0	0	0	**105**	**0.2**
Bird of prey	682	0	0	0	0	**682**	**1.2**
Unknown predator	2413	0	0	0	0	**2413**	**4.3**
	29110	23130	2276	2182	0	**56698**	

5.3.6 Circumstances of predation

There was no marked seasonal difference in the number of incidents of livestock depredation between the cool dry (54. 5%) and hot wet (45.5%) seasons. The month of June had the highest recorded number of attacks, but care must be taken when interpreting these figures because poor record keeping means that more recent predation incidents were probably better remembered.

Forty nine percent of depredation incidents occurred at night, of which the majority were outside the kraal (91.3%) and altogether 88.8% of incidents, both day and night, occurred outside of the kraal. Yet, all respondents indicated that they kept their small stock in enclosures at night and 62.3% of the farmers who owned cattle indicated that they kept their cattle in enclosures at night. However, many respondents subsequently indicated that depredation outside the kraal at night occurred when livestock had not returned to enclosures on these specific nights.

Respondents were asked about common circumstances of depredation events. The majority of respondents confirmed that most incidents happened at night (58.1%) and outside the kraal (86.1%); whereas only 28.4% of respondents perceived depredation to be primarily during the day, and 11.1% respondents found predation to be mostly inside the kraal. Fifty-nine percent of respondents believed livestock loss to be seasonal of which the cool dry season was regarded to be the most common season for predator attacks on livestock (60.0%).

5.3.7 Trends in perceived predation levels

Respondents were asked to describe the level of conflict with predators over the last ten years. Many respondents felt that depredation incidents were increasing (38.0%) (Table 5.6). Respondents were asked to explain why they felt that human-predator conflict was increasing, most were unsure but some felt it was due to an increase in predator abundances. However, three respondents mentioned that an increase in depredation events was due to the lack of grazing resulting in weaker livestock which make for easier prey for predators. To compensate for the poor grazing, many farmers would leave the livestock out at night to have more grazing time. Of the respondents who indicated that conflicts were decreasing (24.1%), improved farm management was reported by five respondents as the reason for changes and two respondents attributed the decrease in conflict to a decrease in predator abundances.

Table 5.6 Summary of the perceived trends in livestock depredation over the last ten years (n = 79). The first column N is the number of respondents and the second column is the percentage of all respondents.

Trends	*N*	Percentage
Increasing	3	38.0%
Decreasing	1	24.1%
Stable	2	27.8%
Unsure	8	10.1%

5.3.8 Predators removed by farmers

Respondents were asked if they had removed (killed or caught) predators in the past ten years. Only seven of respondents admitted to having removed predators. A cheetah had been killed by hunting with domestic dogs, a number of hyenas (numbers unspecified) had been caught in gin (leg-hold) traps and subsequently killed, a number of black-backed jackals and hyenas (numbers unspecified) were killed by hunting with dogs, and a leopard had been shot with permission of the Department of Wildlife and National Parks (DWNP).

5.3.9 Livestock loss

Overall, respondents classified drought (67.5%) as their biggest problem when it came to their livestock farming, followed by predators (60.0%) and diseases (53.8%). However, insufficient grazing was also expressed as a major problem by a number of respondents (47.5%). Other problems encountered by farmers were infertility, poor quality grazing, low yields, unreliable market, theft, snares, veterinary cordon fences and miscarriages. Insufficient grazing was identified as the second most important concern by about two-thirds (66.7%) of respondents who did not express it as the biggest problem.

5.3.10 Knowledge of local predators

Less than half the respondents could correctly identify a cheetah (48.8%). Leopards were correctly identified by 57.5%. Leopards and cheetahs were often confused with each other. By contrast, 77.5% of the respondents could identify lions and 75.0% a black-backed jackal. Wild dogs and brown hyenas could only be identified by 43.8% and 31.3% of respondents, respectively. Overall, correct identification of all eight predators was very poor, with less than half of the respondents capable of identifying all of the common predators. Interestingly, respondents who had scored highly on predator identification were asked where they had learnt to identify these predators. It was found that those with family members who work, or used to work, in game reserves or the like (e.g. captive facilities) had a particularly high score. Fifty four percent of respondents that correctly identified all predators knew someone that worked in a reserve.

5.3.11 Attitudes and perception towards wildlife

Sixty nine percent of respondents believed that cheetahs should be protected in Botswana but despite this, only 38.5% of respondents attached any positive value to the cheetah. Positive values included ecotourism, beauty, employment, and the need for future generations to see cheetahs. Only 20.0% of respondents had a positive attitude towards sharing the land with predators, 59 (73.8%) had a negative response and five respondents (6.3%) were either indifferent or unsure. Despite the negative perception of the coexistence of predators on farmland, the majority of respondents (80.0%) agreed that wildlife should be protected as it is a national resource. In response to the question, “who do you think is responsible for the predator-livestock conflict?” 53 out of the 79 respondents (67.1%) believed it to be the responsibility of the Botswana government. Seven respondents felt it was the responsibility of the owner of the livestock, whilst six respondents held the game reserves responsible. Other parties that were mentioned as accountable were non-governmental organizations (NGO) and everyone, whilst four respondents were unsure. There were mixed opinions regarding the solution to the protection of wildlife. Respondents primarily mentioned translocation as a solution towards the coexistence of predators on farmland (35.0%). Improved farm management was expressed by 20.0% of total responses.

5.3.12 Attitude, knowledge and husbandry Indices

Predictors of Attitude

The mean attitude index score was -0.29 ± 2.27 (range: -4 to 6). A large proportion of respondents (46.3%) had an attitude index of -1 or less and only 8.8% of respondents had an attitude index with a positive value of three or more, indicating an overall negative attitude towards predators on farmlands. The attitude index had a low Cronbach reliability $\alpha = 0.35$ (range: 0.26-0.37). The Cronbach's alpha value is low in relation to the acceptable reliability value of 0.70 (Santos 1999). However, lower thresholds are sometimes used and reported in the literature (Santos 1999). All questions/statements produced similar values so no single question/statement could be removed to improve (increase) the overall reliability of the index. The low alpha value may be the product of a limited number of questions/statements used ($n = 6$) to construct the index (Gliem & Gliem 2003).

A GLM identified 17 out of 256 potential models to best explain the attitude index (ΔAICc value < 2). These models included six predictor variables; knowledge index, husbandry index, distance to the reserve, respondent's position, highest level of education, and primary form of income (Table 5.7). However, the cross-validation GLM (i.e. a GLM that only included the six variables identified above) indicated that no one predictor variable/combination significantly influenced the attitude indices of the respondents.

Table 5.7 The top 17 models and identified variables used to predict the attitude index. Models are arranged in descending orders according to their AICc scores. 'Education' refers to the highest level of education, 'Income' is the main source of family income, "Position" explains the position (owner/employee) held on the cattlepost, 'Distance' describes the kilometres to the boundary of the reserve, 'Husbandry' refers to the Husbandry Index score, and 'Knowledge' is the Knowledge Index score.

Model	**Education**	**Income**	**Position**	**Distance**	**Husbandry**	**Knowledge**	**AICc**	**ΔAICc**	**AICc weight**
77		X	X			X	337.4	0.00	0.038
109		X	X		X	X	337.5	0.05	0.037
101		X			X	X	337.5	0.10	0.036
69		X				X	337.7	0.33	0.032
89			X	X		X	338.1	0.65	0.028
73			X			X	338.2	0.81	0.025
65						X	338.2	0.81	0.025
67	X					X	338.5	1.04	0.023

93		X	X	X		X	338.5	1.12	0.022
75	X		X			X	338.7	1.27	0.020
125		X	X	X	X	X	338.7	1.27	0.020
81				X		X	338.7	1.32	0.020
117		X		X	X	X	339.0	1.54	0.018
91	X		X	X		X	339.0	1.63	0.017
83	X			X		X	339.1	1.73	0.016
85		X		X		X	339.2	1.74	0.016
71	X	X				X	339.3	1.91	0.015

Predictors of Husbandry

The husbandry index scores were generally positive, with a mean score of 3.3 ±3.5 (range -4 to 14). The index had an acceptable internal consistency (Cronbach reliability $\alpha = 0.56$; range = 0.63-0.71). Nine predictor variables were used to test their effects on the husbandry index.

Six models (ΔAICc value < 2) best described the data and these included four of the predictor variables; perceived common circumstances of attack, main form of family income, maximum gap size recorded in the kraal walling, and total number of livestock loss to predators in the previous 18 months (Table 5.8). The 'best' model (model 67) included both circumstances of attack and gap size ($w_i = 0.065$). However, the cross validation GLM indicated that no one predictor variable/combination significantly influenced the husbandry indices of the respondents.

Table 5.8 Summary of the top six models and four predictor variables that best described the dataset for the husbandry index. The models are arranged in descending order based on their AICc values. 'Circumstances of attack' refers to the common circumstances of depredation incidents on livestock, 'Income' is the respondent's main source of income, 'Gap size' is the largest size gap (cm) recorded in the kraal walling, and 'Livestock loss' refers to the total number of livestock loss to predators in the previous 18 months.

Model	**Circumstance of attack**	**Income**	**Gap size**	**Livestock loss**	**AICc**	**ΔAICc**	**AICc weight**
67	X		X		338.1	0.00	0.065
3	X				338.2	0.09	0.062
73		X	X		338.8	0.69	0.046
65			X		339.4	1.28	0.034
201		X	X	X	339.5	1.43	0.032
195	X		X	X	339.9	1.86	0.026

Predictors of Knowledge

The mean knowledge index was 2.76 ± 5.25 (range: -9 to 14) (Cronbach reliability α = 0.69; range = 0.45-0.59). Five models best described the data, with four predictor variables (age of respondents, the highest level of education, the distance of the cattlepost to the reserve boundary, and the number of years respondents had been living in the area) (Table 5.9).

Model 22 had the highest Akaike weight (w_i = 0.118) and included age, distance and livestock loss (Table 5.9). However, the cross validation GLM indicated that no one predictor variable/combination significantly influenced the knowledge indices of the respondents.

Table 5.9 Summary of the five top models and variables that best predicted the knowledge index. The models are in descending AICc order. 'Age' refers to the respondent's age in years, 'Education' is the highest level of education obtained, 'Distance' is the cattlepost's distance (km) from the boundary of the reserve, and 'Years' refers to the number of years a respondent has lived in the area.

Model	Age	Education	Distance	Livestock loss	Years	AICc	ΔAICc	AICc weight
22	X		X	X		470.1	0.00	0.118
21			X	X		470.2	0.09	0.112
6	X		X			471.5	1.46	0.057
54	X		X	X	X	471.9	1.81	0.048
23		X	X	X		471.9	1.86	0.046

5.4 Discussion

Questionnaire interviews can be used to evaluate interactions between humans and predators, providing measurable data that can be quickly and relatively cheaply collected (Holmern & Røskaft 2013). However, caution needs to be exercised when reviewing results as questionnaires rely on information that is subjective and sometimes misleading. For example, the exaggeration of livestock losses and bias towards certain predator species (Graham *et al.* 2005; Holmern & Røskaft 2013). Furthermore, the relatively small dataset used in my study, despite including all available cattleposts, also likely affected the results making it difficult to determine relationships between ecological and social variables.

5.4.1 Livestock husbandry practices

Livestock protective methods in the communal farmland consisted mostly of the kraaling of livestock at night. However, most livestock herds were not fetched in the afternoon and left to return on their own. Furthermore, some farmers did not count their livestock upon return so any strays would sleep out at night. During the day livestock herds were mostly left unattended and unprotected with relatively few farmers employing herders and/or LGDs to protect their livestock. This is despite respondents considering LGDs to be effective. The use of LGDs has been documented as an effective tool to reduce predation (Coppinger *et al.* 1988; Marker *et al.* 2005; Gonzalez *et al.* 2012). Only 16 respondents used LGDs and these were only used to protect small stock, yet a total of 239 domestic dogs (mean and SD: 2.99 ± 2.55 per cattlepost) were owned on cattleposts. The relatively low number of farmers that used LGDs as a protective method suggests that farmers are either not aware of the benefits of using dogs to protect livestock, or do not know how to train a dog to become a livestock guardian (pers. obs.). Livestock farmers in the communal farmlands may therefore benefit from a formal workshop on the use of LGDs including instructions on livestock guard dog training. Furthermore, subsidising of dog food may be an incentive for farmers to adopt this protective method (pers. obs.).

Four main livestock kraal designs were identified and consisted of tightly fitted mopanewood (*Colophospermum mopane*) posts, split-rail fencing, tightly packed acacia branches, or wire fencing. Some farmers also had maternity kraals where young goats and sheep would be kept separately during the day. All kraals that were built with the use of acacia branches had the stems of the branches facing out of the kraal. Additionally, the walling of all kraal designs, particularly for cattle, typically had large gap sizes suggesting that kraals are built to prevent livestock from escaping rather than to prevent predators from entering the kraals (see also Patterson *et al.* 2004). Nonetheless, kraaling livestock at night, regardless of the kraal design, appears to be effective in reducing livestock losses as livestock depredation was mostly recorded when farmers failed to kraal their livestock at night. Similar findings have been recorded in previous studies (Schiess-Meier *et al.* 2007; Kgathi *et al.* 2012; Parker *et al.* 2014). Large predators have been found to avoid close proximity to human settlements (Ramakrishnan *et al.* 1999; Ogada *et al.* 2003; Pettorelli *et al.* 2009). Therefore, keeping livestock at night close to the homestead, regardless of the kraal design, may decrease depredation incidents. Some farmers took added precaution by building fires around the kraals or building some form of roofing on maternity kraals to prevent baboons from entering the enclosures during the day when homesteads were unoccupied. Other protection methods utilised included raised chicken pens to protect chickens from honey badgers. These findings suggest that, where protective measures are used, they are effective but many farmers lack a proactive approach towards the raising of their livestock. Regaining farmers' self responsibility for their livestock may change overall husbandry effort, as active defence and herd attendance are essential measures of animal husbandry (Patterson *et al.* 2004).

Despite the higher value of cattle, husbandry methods were typically less intensive for cattle than for smaller stock. Cattle were mostly left to roam freely and unattended. Furthermore, one farmer indicated that he had not seen his cattle in over two weeks as they had wondered off in search of better grazing. However, despite a lower husbandry effort, cattle did have a lower depredation rate than goats (cattle = 3.7% goats = 13.0%). Some farmers had expressed that because of a lack of grazing they had recently moved their cattle to other areas, whereas other farmers were allowing cattle to graze for longer hours sometimes leaving them to graze out during the night. Thus, the relaxed protective methods observed may have been temporary in reaction to the drought conditions, and more stringent kraaling may ordinarily be used. This might explain the lower depredation of cattle. Another possible explanation is that cattle are typically preyed upon by larger predators such as lions (Patterson *et al.* 2004; Holmern *et al.* 2007; Selebatso *et al.* 2008; Sifuna 2010; Kgathi *et al.* 2012). Thus, the relatively low depredation of cattle may indicate that larger predators (i.e. lions) occur at lower densities on farmlands than smaller predators such as black-backed jackals (Patterson *et al.* 2004). Indeed, lions were mostly described as absent by respondents (47.4%) on communal farmlands. Furthermore, certain livestock types may be more vulnerable than others due to differences in behaviour including herd composition and vigilance (Polisar *et al*. 2003). For instance, some livestock may have less flight capability and weaker defences (Polisar *et al*. 2003).

Accurate record keeping of past incidents of livestock depredation was typically lacking; this was also noticed in a previous study in the Ghanzi District in south-west Botswana (Selebatso *et al.* 2008). In addition, some farmers did not know the exact number of livestock and poultry they owned. Poor record keeping and irregular livestock inspection may cause farmers to unjustly blame predators for livestock losses (pers. obs.). Additionally, it appeared that depredation incidents by larger predators including lions, leopards and cheetahs were more easily remembered than depredation events by baboons, black-backed jackals and hyenas. Respondents would often name incidents by larger predators and only report other incidents when we questioned them on any losses due to other predators. Possible reasons for this are that larger predators may hunt cattle which have a higher value to farmers, hence incidents are more noteworthy, and furthermore, livestock losses by these predators accrue compensation.

5.4.2 Wildlife and predators

Accurate information on wildlife (prey availability) and predator abundances is essential to explain predator-prey interactions (Landa *et al.* 1999; Graham *et al.* 2005), as human predator conflict is often indirectly fuelled by the depletion of wild prey from poaching and competition with livestock (Graham *et al.* 2005; Winterbach *et al.* 2014). There is a lack of data on the abundances of wildlife and predators occurring in the communal farmlands adjacent to NOTUGRE, but during my extensive fieldwork in these areas I rarely saw wildlife. Respondents were asked to provide estimations on wildlife occurrences and frequencies of sightings. Larger species, including eland, zebra, wildebeest and kudu were generally considered to be absent. Only smaller prey species including steenbok, scrub hare and helmeted guinea fowl were described as being common. However, these are perceived abundances which are based on a very subjective evaluation. Significantly, livestock density was calculated at approximately 20.7 livestock/km². An overexploitation of wildlife (through poaching), coupled with high livestock density and a corresponding increase in competition for natural resources (food and water), can reduce the density of wildlife, particularly of large prey (> 60 kg), outside of protected areas (Mishra *et al.* 2003; Graham *et al.* 2005; Winterbach *et al.* 2014). Predators will prey upon wild prey species in preference to domestic livestock, but where the prey base is absent or limited, predators may resort to killing domestic livestock (Landa *et al.* 1999; Patterson *et al.* 2004; Graham *et al.* 2005; Schiess- Meier *et al.* 2007; Winterbach *et al.* 2014). Livestock losses are not related to predator density, but are rather a function of livestock availability (Landa *et al.* 1999; Graham *et al.* 2005). Therefore, reducing predator abundances, in anything less than a radical eradication of isolated populations, is unlikely to resolve conflict (Landa *et al.* 1999; Graham *et al.* 2005). Predators are more likely to prey on livestock where livestock occurs at higher densities than wild prey (Landa *et al.* 1999; Polisar *et al.* 2003; Graham *et al.* 2005; Winterbach *et al.* 2014). However, where livestock is well looked after, including kraaling during the night and guarding during the day, and the natural prey base is not depleted, large predators will prey upon wild prey even when livestock is more abundant (Marker *et al.* 2003d; Ogara *et al.* 2010; Winterbach *et al.* 2014). Conserving natural prey should not be overlooked when attempting to reduce livestock depredation in the context of large predator conservation (Polisar *et al.* 2003; Mishra *et al.* 2003; Clarke 2012). The complex ecological interactions require a multi-species and ecosystem management, thus it is important to also consider the quality of the habitat as this might affect prey availability (Graham *et al.* 2005). The severe overgrazing observed in the communal farmlands is likely to negatively affect the abundance of wild prey species.

5.4.3 Level of conflict

Livestock losses were reported to be caused most often by hyenas, followed by black-backed jackals and baboons. This was reflected by the respondent’s ranking of predator importance. Hyenas were also reported to be the most substantial damage-causing predator in the Ngamiland District in northern Botswana (Kgathi *et al.* 2012), and near the Serengeti National Park in Tanzania (Holmern *et al.* 2007). In a study conducted in Zimbabwe, baboons were reported to contribute to the majority of goat and sheep predation (Butler 2000). Leopards were not important predators in my study, contributing to 7.3% of all livestock losses and 8.7% of total economic value. In contrast, leopards were ranked as themost frequent predator in studies conducted in the Ghanzi District in south-west Botswana (Selebatso *et al.* 2008) and the Okavango Delta region in the Ngamiland District in northern Botswana (Sifuna 2010). In terms of economic loss, lions were found to be the third most important predator, particularly for cattle. A number of studies also found lions to be important predators of cattle (Patterson *et al.* 2004; Holmern *et al.* 2007; Selebatso *et al*. 2008; Sifuna 2010; Kgathi *et al.* 2012). However, these findings may have been associated with species that accrue compensation, particularly for studies that obtained data from DWNP Problem Animal reports (Selebatso *et al.* 2008). Cheetahs attacked mostly smaller prey (goats) with only one recorded attack on a calf, with a total contribution of only 4.5% of all livestock losses to predators, which represented 3.4% of the total economic value of livestock depredation.

The observed high depredation by hyenas and black-backed jackals may suggest a higher abundance of these species (Patterson *et al.* 2004; Yirga *et al.* 2014). However, both species have distinctive calls and so are more easily detected and identified than other predators (Skinner & Chimimba 2005). Nonetheless, they are both opportunistic feeders and have behavioural plasticity and so may prey upon vulnerable livestock and scavenge from livestock carcasses (Hall-Martin & Botha 1980; Yirga *et al.* 2014). Due to their ecological flexibility and behavioural plasticity both of these species are more likely to adapt to anthropogenic landscapes and therefore be important predators of livestock (Holmern *et al.* 2007).

The depredation impact (percentage lost) on the livestock in my study constitutes a significant proportion of the total livestock owned (~9.8%) in comparison to other studies (Graham *et al.* 2005). In a comprehensive global study on human-predator conflicts, Graham *et al.* (2005) found livestock loss to range between 0.02 – 2.6% of all livestock owned. Sixteen respondents in my study reported livestock depredation that exceeded 25% of their total number of livestock. This economic impact can be substantial for poor rural subsistence farmers that may only own a few livestock (Mishra *et al.* 2003). For instance, loss of one sheep or goat may represent a loss of one month's pension for a cattlepost resident. This high economic impact may reduce tolerance towards predators and provoke retaliatory persecutions (Sifuna 2010; Lindsey *et al.* 2013; Chattha *et al.* 2013). This is particularly so where livestock provides the only means of livelihood (Dickman 2010).

Tolerance of livestock depredation differed with the different livestock types. Tolerance appears to be associated to the social value of the livestock, thus the loss of a cow was considered to be substantial however very little social and/or monetary value was attached to donkeys and their losses were often tolerated and left unreported. Donkeys were the most depredated livestock type relative to the number owned (21%), whereas cattle were the least (3.7%), yet because of the social value attached with cattle it is likely that farmers have a lower tolerance towards predators attacking cattle. Thus, addressing the problem of depredation of more valuable livestock should be a priority when implementing livestock mitigation measures.

Interestingly, the loss of chickens was often considered to be important and respondents often commented on the problem of honey badgers and other small predators. Most studies have focused on the human-large predator conflict yet small- (average body weight <7kg) and medium-sized predators (7-25kg) may prey upon poultry and even young kids and lambs (Holmern & Røskaft 2013). Honey badgers, African wild cats, and birds of prey were often reported as important predators by farmers. Furthermore, baboons and black-backed jackals were also reported as important culprits in poultry loss. Poultry can have an important nutritional and financial value for rural farmers (Holmern & Røskaft 2013).

Livestock diseases and poor nutrition due to drought were most frequently identified as the biggest problems faced by livestock farmers. All interviews were conducted over the dry season and over a drought, hence the results may have been influenced by prevailing conditions. Unfortunately, due to poor record keeping, I could not quantify the value of these losses. However, Graham *et al.* (2005) found that many studies evaluate livestock losses to other causes than predators to be proportionally more financially damaging (Graham *et al.* 2005). Predation may also mask underlying causes such as poor husbandry including poor diet and health (Graham *et al.* 2005). Nonetheless, other causes of livestock loss are often not considered by farmers and/or are more tolerated.

The presence of a predator does not prove livestock depredation. Predators are sometimes blamed for missing or stolen livestock (Graham *et al.* 2005). This was also the case in my study where a number of respondents held predators responsible for missing livestock. Poor husbandry practices predispose such behaviour as carcasses are seldom found when livestock is left unattended and thus predators are blamed with little or no proof. It is therefore difficult to conclude what percentage of livestock losses is positively a result of depredation without intensive monitoring.

5.4.5 Trends

Many respondents felt that livestock losses to predators have been increasing in the last ten years. The reason was often said to be the growing numbers of predators although some farmers expressed that poor grazing quality and starving livestock may have also lead to predators preying upon the weaker animals. However, increased predation incidents may be as a result of a depleted prey base and increasing livestock densities. It is also possible that respondents expressed that the conflict is deteriorating to emphasise the severity of the problem (Marker *et al.* 2003c). “Hyper-awareness” is also common; this is where respondents exaggerate their losses intentionally or unintentionally even where they may not have personally experienced wildlife conflicts (Dickman 2010). Perceptions of damage causing predators may come about from only one incident experienced by a community member (Dickman 2010). For instance, during my interviews a respondent was particularly unhappy with wild dogs despite never loosing livestock to wild dogs.

5.4.6 Farm management recommendations

The level of human-predator conflict on communal farmlands appears to be high; livestock losses are extensive and persecution of large predators’ both outside and within the reserve may have severe consequences on predator populations, particularly on the relatively small cheetah population (see Chapters 3 and 4). Present livestock husbandry measures appear to be insufficient for acceptable and tolerable levels of livestock losses. Improving current farm management and animal husbandry practices, including implementing a proactive attitude such as daily record keeping, fetching livestock from pastures and ensuring all livestock has returned and is kraaled at night, will not only reduce incidents of livestock loss due to predators (Graham *et al.* 2005), but more importantly, the loss of livestock by theft, snaring, diseases and starvation should also decrease as farmers will have the opportunity to identify any sick or injured animal (Schiess-Meier *et al.* 2007).

Good husbandry practices; livestock accompanied by a herder during the day, kraaling livestock at night with LGDs reduces livestock depredation and may, in the long term, prevent predators from becoming habitual livestock hunters (Marker 2002; Ogada *et al.* 2003). Kraals need to be predator proofed and built away from dense bushes and in close proximity to active homesteads (Ogada *et al.* 2003). Losses can further be reduced by burning fires around the kraals to deter predators at night (Kgathi *et al.* 2012), synchronized livestock breeding seasons and using calving kraals that are well protected (i.e. roofing) and close to human habitation (Marker 2002; Polisar *et al*. 2003), and stocking certain breeds of cattle and goats that are less vulnerable to predation than others (Landa *et al.* 1999; Polisar *et al*. 2003). Furthermore, the livestock needs to be in healthy condition and well fed, this may require reducing the density of livestock on the already overgrazed land and moving the livestock to more arable land during drought periods. Seasonal management of livestock may further reduce frequency of predator attacks that are elevated in the dry season. The farming community can also assist in restoring wild prey populations by ensuring there is sufficient available forage and reducing poaching.

5.4.7 Management implications

Over an extended period of time (from the 1890s until 1960s) there was widespread eradication of all large carnivores in the Tuli Block (Lind 1974) as they were seen as vermin by livestock farmers establishing farms. Consequently, large carnivores have, for the most part, been extirpated from farmlands within the Tuli Conservation zone (Winterbach *et al.* 2014). However, with the establishment of game reserves, large carnivore populations have shown some recovery in numbers and re-occupation of former ranges has taken place (McKenzie 1990). Due to the prolonged absence of large carnivores, most traditional husbandry practices have been abandoned over time (Kgathi *et al.* 2012). Indeed, livestock guarding in the rural communal farmlands is limited and farmers lack a proactive approach towards the raising of their livestock. Conflicts between people and predators are emerging and growing in regions that are experiencing recovering predator populations after extended periods of local extinction. But there may be resistance among farmers in readopting some of the traditional husbandry practices as they are potentially costly (i.e. employing a herder) and require willingness to a change in lifestyle (Ogada *et al.* 2003). Children were commonly used as herders in the past but are now required to attend school (Kgathi *et al.* 2012). Technical assistance and economic support, such as subsidy of husbandry practices may encourage farmers to change their farm management practices and reduce depredation rates, possibly providing the first step towards mitigating the HWC. Alternatively the DWNP could enforce the use of responsible farm management (Klein 2007).

5.4.8 Aspects of attitude, knowledge and husbandry

Attitude is considered to be an important aspect of conflict mitigation efforts, with the prevalent assumption that hostility is directly affected by the level of predation (Dickman 2010). No set of factors best explained the attitude, husbandry or knowledge of respondents living alongside predators, although the knowledge index was identified as an important factor in shaping respondents' attitudes. Previous studies have reported that education and knowledge are important drivers of attitudes and encourage farmers to be involved in the planning and decision-making concerning the management of large predators beyond protected areas (Bath 1998; Ericsson & Heberlein 2003; Røskaft *et al.* 2007; Selebatso *et al*. 2008).

Respondents expressed a negative attitude overall towards the conservation of large predators. But despite livestock forming an important source of income and food, neither economic dependency nor the extent of livestock loss influenced the attitudes of respondents. Attitude is not only shaped by human-wildlife interactions and personal experiences, but it may also be a product of social factors and human-human conflict (Dickman 2010). Interactions between people and authorities can play a substantial role in human-predator conflict and is often overlooked (Dickman 2010). Thus, understanding and improving the relationship between the local people and conservation bodies such as the DWNP and NOTUGRE is equally important to effectively mitigate conflict. If residents have had a negative experience, they may view the reserve or local authorities with a negative attitude which may lead to negative attitudes towards wildlife conservation. Attitudes towards local authorities (DWNP and Botswana government) and NOTUGRE were not investigated in this study but some respondents clearly demonstrated their unhappiness with either the reserve or the local wildlife authorities. Thus, I feel that improving these relationships is a critical aspect towards shaping more positive attitudes and should be investigated further.

5.4.9 Compensation implications

Compensation schemes have been implemented in Botswana in an effort to reduce HWC by increasing tolerance for losses and reducing retaliatory killing of damage-causing wildlife (Bulte & Rondeau 2005; Jackson *et al.* 2008; Selebatso *et al.* 2008; Kgathi *et al.* 2012). Furthermore, it has been suggested that reports from compensation schemes can be used to document the current conflict as farmers are more likely to report livestock losses if they have financial incentives (Klein 2007; Selebatso *et al.* 2008). However, critics have argued that compensation schemes are inefficient in reducing conflict and may even encourage farmers to relax their protective measures (Bulte & Rondeau 2005; Klein 2007; Clarke 2012; Kgathi *et al.* 2012). This is apparent on many cattleposts located outside or within NOTUGRE where the blame for human-predator conflict has shifted towards the government body and livestock protection and care was mostly believed to be the responsibility of the government. Compensation schemes are often inefficient due to a number of challenges associated with implementation, including a high financial budget and man power required to process the claims (Jackson *et al.* 2008; Kgathi *et al.* 2012). The government is also committed to continuing this program indefinitely. Farmers tend to only report incidents which accrue financial compensation, consequently the information gathered from Problem Animal Control (PAC) reports does not necessarily give an accurate picture of the predator conflict as the dataset is invariably biased in terms of predator species and is likely to under estimate the extent of livestock depredation (Landa *et al.* 1999; Schiess-Meier *et al.* 2007). Assisting farmers to protect their livestock is believed to be a better solution (Clarke 2012). In 2009, the Botswana Government updated their compensation policy to include the requirement of adherence to certain farming management practices (herding of livestock during the day and enclosing the livestock into well-constructed kraals at night) to avoid potential moral hazards that may arise from negligent farmers with poor livestock husbandry practices (Bulte & Rondeau 2005; Kgathi *et al.* 2012).

In my study, most respondents had suffered livestock losses to predators (83.8%), yet only about half had previously contacted a wildlife officer. Incidents of livestock depredation were not always reported to wildlife officers, particularly if it was damage done by a hyena as that would not warrant compensation under Botswanan legislation. Some respondents expressed their dissatisfaction with the compensation policy. The current compensation rate for livestock loss due to predators is approximately 35% of the market value of the livestock (Kgathi *et al.* 2012). Therefore, it is unlikely that the compensation scheme is effective in terms of alleviating the human-predator conflict. In addition, the DWNP is responsible for the implementation of laws against the illegal killing of predators, however these are difficult to enforce due to the limited man power available and large distances involved (Klein 2007).

The success of compensation schemes relies on a streamlined, adequate, and efficient system. An incentive program, where the farmers are involved and able to implement decisions within the community, may gain the support of the community for sustainable coexistence between farmers and predators (Clarke 2012). Mishra *et al.* (2003) designed an incentivised program, a locally managed communal insurance program, where farmers contribute monthly premiums for their livestock in a communal insurance fund to offset the costs of livestock losses (Mishra *et al.* 2003). A similar system could be adopted in the communal farmlands bordering onto NOTUGRE. The program appoints local community members to supervise the implementation of the insurance compensation scheme and regulations of the funds are discussed between the community council and the government body (Mishra *et al.* 2003). Initially, the Government and NGOs can help contribute funds into this cooperative fund until it is self-sustaining (Mishra *et al.* 2003). Incentives may be provided to encourage good livestock husbandry by rewarding farmers that have had the least annual number of livestock losses; producing realistic rates of compensation at 100% the value of the livestock; and discouraging false compensation claims (Mishra *et al.* 2003).

5.4.10 Outreach

The general consensus among respondents for resolving conflict was to translocate large predators out of farmlands. Furthermore, respondents often supported the conservation of wildlife but only within protected areas. A similar predisposition was found in a questionnaire survey in the Ghanzi District, where significantly fewer farmers supported the conservation of cheetahs outside protected areas (Selebatso *et al*. 2008). Promoting the value of wildlife in farmland ecosystems can increase conservation awareness (Marker *et al.* 2003a). Equally, understanding the impact and consequences of persecution, particularly indiscriminate killing by the use of wire snares and poison, is crucial in the preservation of biodiversity. Environmental education programs provide a platform to explain the current effect of predator persecution and the successful non-lethal methods available to reduce the loss of livestock. However, the success of this program relies on complete transparency from conservation authorities where the purpose of the program is clearly stipulated from the outset otherwise a negative attitude may be formed from false, negative and incorrect information given. This is best achieved by providing specific knowledge such that local communities can make informed decisions. The community may further benefit from information on local predator species including techniques to identify culpable predators in an event of predator loss.

Although environmental awareness can improve the overall understanding of the importance of wildlife, rural farmers may have other priorities (Clarke 2012). Sustainable use of the land for long term benefit is not necessarily a priority, many farmers live day-to-day and there is little incentive to protect wildlife which does not give direct financial benefit (Mishra *et al.* 2003; Clarke 2012). Economic incentives for the conservation of wildlife on communal farmland may increase the value of wildlife, such as through the well managed and sustainable consumptive use of wildlife, and can result in positive attitudes (Klein 2007; Sifuna 2010). It is important that the local people's needs and rights are taken into account (Clarke 2012). Economic gains such as ecotourism and hunting can increase the value of wildlife and hence increase wildlife tolerance and attitudes towards the coexistence of predators on farmlands (Mishra *et al.* 2003; Klein 2007; Sifuna 2010). Increasing wildlife numbers by the banning of hunting has the reverse effect; Kenya is a prime example of failure, losing 60-70% of all its wildlife since the ban of hunting and consumptive use of wildlife in 1977 (Clarke 2012). Furthermore, legalising consumptive and sustainable harvest and giving authorization for communities to jointly manage their wildlife may reduce poaching which largely comes from communities that border onto reserves (Sifuna 2010). The survival of wildlife relies upon the support of local communities and consumptive use of wildlife is likely to encourage this support (Sifuna 2010).

Local communities require ownership of natural resources and involvement in decision-making regarding wildlife management (Bath 1998). However, successful community run concessions require a flawless operation that is free of corruption and greed so that income generated from wildlife benefits those affected (Clarke 2012). A successful outcome requires interest and dedication on the part of the community, but this might be difficult to achieve as the average farmer does not have the desire to work harder and has few ambitions (Clarke 2012). However, increasing their appreciation for wildlife could gain their support for conservation initiatives (Hudenko 2012). Positive encounters with wildlife can evoke a positive emotional response and affection which can positively change the attitude towards wildlife (Røskaft *et al.* 2007; Hudenko 2012; Heberlein 2012). Educational programs such as Children in the Wilderness (CITW) take children from local communities to neighbouring lodges in protected areas where the children not only learn about the importance of the natural environment but are also taken on wildlife viewing drives where they have a chance to see and experience their natural heritage, inspiring them to become custodians of the environment.

5.5 Conclusion

Livestock losses experienced by farmers in farmlands adjacent to and within NOTUGRE appear to be relatively high compared to previous studies, but may be a consequence of the lack of proactive livestock protection measures. Farm management training that includes preventative measures for livestock depredation, correct techniques to identify the predators responsible as well as overall improved livestock husbandry would benefit rural subsistence farmers. Farmers need to gain self-accountability and responsibility for their livestock, which requires them to better protect their livestock from predators and improve current livestock husbandry practices. The current compensation scheme was initiated as a measure to mitigate the human-predator conflict however it does not appear to have resolved the problem and may even have shifted responsibility. In so doing, the wildlife authorities are perceived to be accountable for the conflict. Conflict mitigation plans may benefit from a locally managed communal insurance program that is implemented by the community in collaboration with the DWNP; improving self-responsibility as well as the relationship between local communities and wildlife authorities. Improved co-operation may also be achieved by organising farmers’ meetings to address concerns in the farming community and the DWNP assisting with infrastructural support. HWC mitigation and the coexistence of predators and necessitates a more positive attitude towards the conservation of predators (Bath 1998).

Human-predator conflict cannot be resolved by reducing the losses of livestock and understanding the socio-economic environment is crucial to the design and implementation of successful conservation plans (Dickman 2010). Conflict resolution requires a multi-disciplinary approach that is specific to the area and includes all stakeholders (Dickman 2010).

CHAPTER 6

SYNTHESIS AND CONCLUSION

Prior to my study there had been no research conducted on the cheetah (*Acinonyx jubatus*) population of the Northern Tuli Game Reserve (NOTUGRE), Botswana. My study aimed to estimate the abundance and status of this population. I used two non-invasive techniques to provide population estimates. Additionally, I determined the attitudes towards wildlife of rural farmers living within and adjacent to the reserve and quantified the level of depredation on livestock to understand the possible threats faced by cheetahs and other large predators on communal farmlands.

6.1 Camera trapping: a monitoring technique for cheetahs

I used camera trapping as a method to estimate population density. The findings assisted in developing camera trapping as a tool for deriving population estimates for cheetahs; a species that occurs at low population densities (Caro 1994) and has relatively unpredictable movements (see Chapter 3). Camera trapping is an affordable, repeatable and non-invasive method that can be used to monitor cheetah populations where scent marking posts are known and accessible. The method was refined from a method used in previous studies conducted in north-central Namibia and the Thambazimbi district of the Limpopo Province in South Africa (Marker *et al.* 2008b; Marnewick *et al.* 2008). When using scent marking posts, it is important to consider that there may be variation in individual detectability (Otis *et al.* 1978). Male cheetahs may use scent marking posts more frequently than females (Marnewick *et al.* 2006), therefore females may be inadequately represented within the dataset both with regards to the number of captures and identified individuals (Marker 2002; Marnewick *et al.* 2006; Marker *et al.* 2008b). Differences in detection probabilities may be accounted for, where sample size permits, by incorporating sex-specific encounter rates into Spatially Explicit Capture Recapture (SECR) models (O'Connell *et al.* 2011). Nevertheless, a possible drawback of using cheetah-specific camera trapping sites is that not all members of a predator guild can be simultaneously surveyed. Hence, the method cannot be used for monitoring multiple species.

The recommended maximum number of days to maintain population closure when conducting capture-recapture studies on large felids is 90 days (Karanth & Nichols 2002). However, my study found a high latency to first cheetah detection (range: 9-85 days) and a relatively low sample size (n = 18 independent cheetah capture events) after this initial 90 day survey. Extending the survey length (n = 130 days) increased the sample size (n = 31 independent cheetah captures) and the robustness of the density estimates. However, a consequence of the long survey period is that population closure may have been violated (Foster & Harmsen 2012). Furthermore, no significant differences were found in the density estimates achieved. Increasing the surveyed area may reduce the effect of a high latency to first detection as the spatial scale of my study area (±240 km²) may have been too small to incorporate sufficient home ranges of cheetahs (see Chapter 4: distribution and home range). Cheetahs have very large home ranges (Marker *et al.* 2008a) and I would therefore recommend that when designing camera trap surveys for cheetahs, the area containing the trap array should be large enough to incorporate sufficient home ranges. Maffei & Noss (2008) recommend that the surveyed area should be at least three to four times the average home range for the target species for that specific site. However, this requires a prior knowledge of the home range size for cheetahs in that specific area, as the size of cheetahs' home ranges varies substantially between geographic locations due to differences in habitat type and prey density (Caro 1994; Gros *et al.* 1996; Broomhall *et al.* 2003). Additionally, the spacing between camera traps may be increased as to optimize trap spacing and accommodate for the relatively large surveyed area required. Dillon & Kelly (2007) recommend at least two camera traps per average home range. Furthermore, in place of the Adjacent Block method, the entire area should be surveyed throughout the sampling period with traps set at half the density and moved to their new location within the same area after half of the sampling period as suggested by Foster & Harmsen (2012). This approach reduces the confounding effect of space and time associated with the adjacent block method (Foster & Harmsen 2012). Deploying a single camera trap unit at scent marking posts increased the total area that could be surveyed, although this method can decrease capture probability due to variable trap effort from malfunctioning equipment and missed individuals. Nonetheless, I feel that this approach was successful as camera traps were set to either take short video clips or a burst of three images during a single trigger event. Additionally, the scent marking posts functioned as a natural lure and thus individuals would usually investigate the scent marking posts for a few minutes increasing the chances of multiple images.

6.2 Citizen science in cheetah research

Citizen science, whereby volunteers assist with data collection, has become increasingly important in ecological research (Silvertown 2009) as not only can a large amount of data be quickly collected but it can also create awareness and a sense of conservation stewardship

(Silvertown 2009; Marnewick & Davies-Mostert 2012). Photographic survey methods employ citizen scientists to collect information on the population ecology of rare, elusive and individually identifiable species (Silvertown 2009). Cheetah specific photographic surveys have been successfully implemented in a number of national parks, including the Kruger National Park and the Kgalagadi Transfrontier Park in South Africa (Bowland & Mills 1994; Knight 1999; Kemp & Mills 2005; Lindsey *et al.* 2009; Marnewick & Davies-Mostert 2012; Marnewick *et al.* 2014). In this study I evaluated the use of a photographic survey to estimate the minimum number and status of cheetahs in a private game reserve that receives fewer visitors annually than National Parks. I also collected older photographs to assess differences in numbers over time, and to provide estimates of age and relations of frequently sighted individuals. Digital cameras are increasingly accessible and widely used, this is ideal for photographic surveys as photographs retain the date and time at which they were taken. In addition, photographs taken on smart phones or cameras with built in GPS's can also record the physical location of captures. Thus, data collected by the public can be validated by the accompanying metadata (data specific to each photograph). A possible draw back to the method is that the survey relies on incidental photographic records, thus photographs are collected opportunistically and so the frequency of sightings cannot be used to assess space use (e.g. habitat preference) and density as the location of sightings are invariably biased to areas with higher tourism activity. Nonetheless, this method can be used as an ongoing collection of photographs to monitor changes in population sizes. Additionally, photographic surveys can provide baseline data on the number of cheetahs in a reserve, their distribution and demography.

My study found that there was a marked difference observed in the sample effort between the wet and dry season. More photographs were received from the cool dry season compared to the wet season. The cool dry season marks the peak tourism period as well as the landscape typically being more open which makes it easier to find cheetahs. It is therefore recommended to carry out photographic surveys in the cool dry period to increase the total sample size.

Camera traps have become increasingly accessible to the general public with a number of private properties utilising these for recreational purposes (pers. obs.). My study demonstrates the potential usefulness of camera traps as an additional tool for photographic surveys. The majority (89%) of photographic entries submitted from South African farmlands were taken by camera traps. This becomes particularly important where surveys are carried out outside of

National Parks or tourism-focused game reserves where cheetahs and other large carnivores may be skittish and elusive.

In my study, I used the software program Adobe Photoshop Lightroom to manage all photographic data, this software was developed for professional photographers to manage, catalogue and edit large numbers of digital images. It is, therefore, ideal for camera trapping and photographic survey data which typically have large volumes of photographs. I would recommend making use of this program to assist with the identification process associated with camera trapping and photographic surveys.

6.3 Conservation status of cheetahs in NOTUGRE

Although the cheetah has received a high amount of research and monitoring attention, studies have been geographically biased to populations in Tanzania and Namibia and the species is nevertheless still identified as a species of concern with high risk status (Ray *et al.* 2005). Botswana supports a significant number of free-roaming cheetahs and in an effort to conserve the species, a ban on hunting cheetahs has been in place since 1968 (Klein 2007). During this time, cheetahs could only be killed in defence of livestock. In 2000, a memorandum was passed banning all killing of cheetahs (Klein 2007). However, the repercussions for killing a cheetah, a 1000 Botswana Pula (BWP) fine (~US$ 100) or one year in prison, may not be sufficient to discourage offenders (Klein 2007). Furthermore, about half of the cheetahs in Botswana are estimated to range on unprotected farmlands where habitats are undergoing degradation and the species may face persecution in retribution to perceived or actual livestock depredation (Winterbach *et al.* 2014). Illegal trade of cheetahs, particularly sub-adults and cubs, is also a cause of concern (Klein 2007). It is estimated that between 50 and 60 cheetahs are illegally removed from Botswana annually (Klein 2007). The survival of cheetahs in Botswana therefore appears precarious.

My findings suggest that cheetahs in NOTUGRE have a low population density and are possibly undergoing a population decline. While a recovering lion population may contribute to this decline (Laurenson 1994; Durant *et al.* 2004, Snyman *et al.* 2014), it might also be the result of persecution as a result of conflict with livestock farmers outside NOTUGRE. Livestock farmers whom I interviewed generally had a low tolerance for predators on farmlands. Additionally, the results of my study suggest a high total livestock loss due to predators on communal farmlands in comparison to other human-predator conflict studies (Graham *et al.* 2005). This may be attributed to the low abundance of natural prey, particularly larger species which are believed to be mostly absent. The low wild prey abundance may be the result of poaching but may also be attributed to habitat degradation on account of overstocking and poor livestock management, thereby reducing the overall carrying capacity of the land (Klein 2007). A better understanding of the density of the natural prey base is, therefore, required. Nonetheless, a lack of a proactive approach towards the raising of livestock was found to be the primary cause for livestock depredation. Responsible farm management should be enforced (Klein 2007), this requires farmers to regain self-responsibility for their livestock by improving current livestock husbandry practices. The communities may benefit from an incentivised program such as a locally managed communal insurance program (Mishra *et al.* 2003).

Although the high livestock loss to predators is a cause for concern as it may fuel human predator conflict (Graham *et al.* 2005), a negative attitude towards predators in my study was not related to livestock depredation. Farmers had an overall negative attitude towards conservation of large predators on farmlands, but this was not related to economic losses, knowledge or other demographic variables such as age or education, as was found in previous studies (Bath 1998; Ericsson & Heberlein 2003; Røskaft *et al.* 2007; Selebatso *et al*. 2008). I suggest that attitudes may be the product of social factors and human-human conflict rather than an economic loss (Dickman 2010). Addressing human-human conflict and promoting an emotional affiliation towards predators may, therefore, play a greater role in conflict resolution than reducing livestock losses. Positive emotions and increasing appreciation for wildlife can be achieved by the continued education of children (such as by the Children in the Wilderness program) and the development of educational programs for local residents including exposing locals to positive experiences with wildlife. Interest in the conservation of wildlife may be achieved by increasing general awareness of the status of large predators but also by the potential financial returns (Marker *et al.* 2003a).

Cheetahs do not hold any value to most farmers living on communal farmlands (see Chapter 5). Implementing sustainable utilisation could give value to the wildlife and increase tolerance towards predators and thereby their conservation (Klein 2007; Sifuna 2010). Consumptive use of wildlife, however, requires accurate estimates of population densities in order to determine the appropriate offtake (Klein 2007). Options for sustainable wildlife utilisation should be investigated along with alternative livelihoods to livestock-keeping which may benefit communities from coexisting with predators, including ecotourism, veld products, predator friendly meat, and honey production (Klein 2007).

The absence of documented cheetah movement across the South African boundary is also of concern, but may be a result of the limited number (n = 27) of photographic entries received from South Africa. The persistence of genetic connectivity between sub-populations is essential for the viability of the population in NOTUGRE, which will depend upon the level of persecution as well as available movement corridors (free from human disturbances). It is therefore imperative that conservation efforts incorporate the neighbouring farmlands including those in neighbouring countries. However, policy and legislation varies across the three states (Purchase *et al.* 2007). Botswana has listed the cheetah as protected and it cannot be destroyed under any circumstances (Klein 2007). In Zimbabwe, the cheetah is protected but can be killed with a permit from the Wildlife Management Authority (Purchase *et al.* 2007). South Africa has complex legislation with each province providing its own regulations, however the cheetah is protected to some degree and a permit is required to remove or kill an animal (Purchase *et al.* 2007). Further research on conflict with cheetahs in neighbouring countries is imperative and the creation, or maintenance of corridors to promote gene flow should be incorporated in management considerations and policies. It is essential that government authorities are involved in these decisions as they have the authority to implement recommendations both at management and policy levels. Further research should investigate whether links between these sub-populations exist, and identify potential movement corridors between protected areas. This could be achieved by investing more research efforts in neighbouring countries through an ongoing photographic census particularly promoting the use of camera traps. Alternatively, movement of cheetahs could be monitored by the use of satellite collars and genetic sampling could be used to determine relatedness and hence the level of gene flow between South African, Zimbabwean and Botswana cheetahs. Genetic sampling could further help determine whether there is more than one genetically distinguishable population within the samples.

6.4 Conclusion

Cheetahs are challenging mammal species to study as they have large home ranges and occur at low population densities (Gros 1998). This means that a population census has to be carried out over a large area, and despite high sample effort, sample sizes are invariably low. Demographically open populations, like NOTUGRE, are particularly challenging to monitor as movement in and out of the study area is unknown and the population may straddle different properties. Despite these challenges, I believe that my study produced valuable information for the conservation and management of cheetahs in NOTUGRE, providing a better understanding of local cheetah population size, status, and distribution in an area which had previously not been researched; offering a baseline for future studies. My study further provides valuable information on monitoring techniques for future research on cheetahs and other large predators which occur at low population densities.

The cheetah is clearly a species of great concern with an elevated risk status and extensive range loss, and therefore requires dedicated conservation efforts to prevent local extinction (Ray *et al.* 2005). The cheetah is one of the most charismatic flagship species with substantial economic and aesthetic value for the ecotourism industry and tourism-financed conservation areas in Botswana. Furthermore, large predators, such as cheetahs, are key components of ecosystems with flagship status and serve as an important umbrella species for the conservation of biodiversity (Ray *et al.* 2005). Through the study of the predators of an ecosystem the ecosystem as a whole is being studied, consequently the population status of a top predator may serve as an indicator of overall ecosystem function and productivity (Packer *et al.* 2003). Conservation actions directed towards large carnivore species, therefore, are expected to have the greatest impact on overall ecosystem conservation (Buk & Marnewick 2010; Macdonald *et al.* 2010b).

Dusk settles over Mashatu on my last day of field work.

Photo: Eléanor Brassine

REFERENCES

AKAIKE, H. 1974. A new look at statistical model identification. *IEEE Transactions on Automatic Control* **19:** 716–723.

ALEXANDER, G.J. 1984. A preliminary investigation into the relationships between geology, soils and vegetation in the eastern Tuli Block, Botswana. BSc (Hons) thesis. University of Natal. Durban.

ANDRESEN, L., EVERATT, K.T. & SOMERS, M.J. 2014. Use of site occupancy models for targeted monitoring of the cheetah. *Journal of Zoology* **292:** 212–220.

BALINSKY B.I. 1962. Patterns of animal distribution on the African Continent (summing-up talk). Annals of the Cape Provincial Museums. *Natural History* **2:** 299-310.

BALME, G.A., HUNTER, L.T.B. & SLOTOW, R. 2009. Evaluating methods for counting cryptic carnivores. *The Journal of Wildlife Management* **73(3):** 433-441.

BALME, G.A., BATCHELOR, A., DE WORONIN BRITZ, N., SEYMOUR, G., GROVER, M., HES, L., MACDONALD, D.W. & HUNTER, L.T.B. 2012. Reproductive success of female leopards *Panthera pardus*: the importance of top-down processes. *Mammal Review* **43:** 221–237.

BARTLAM-BROOKS, H.L.A., BONYONGO, M.C. & HARRIS, S. 2011. Will reconnecting ecosystems allow long-term distance mammal migrations to resume? A case study of a zebra *Equus burchelli* migration in Botswana. *Oryx* **45:** 210-216.

BASHIR, S., DALY, B., DURANT, S.M., FÖRSTER, H., GRISHAM, J., MARKER, L., WILSON, K. & FRIEDMANN, Y. 2004. *Global Cheetah (Acinonyx jubatus) Monitoring Workshop*. Final workshop report. Conservation Breeding Specialist Group (SSC/IUCN). Endangered Wildlife Trust.

BATH, A.J. 1998. The role of human dimensions in wildlife resource research in wildlife management. *Ursus* **10:** 349–355.

BISSETT, C. 2004. The feeding ecology, habitat selection and hunting behaviour of re-introduced cheetah on Kwandwe Private Game Reserve, Eastern Cape Province. MSc thesis. Rhodes University. Grahamstown.

BISSETT, C. & BERNARD, R.T.F. 2007. Habitat selection and feeding ecology of the cheetah (*Acinonyx jubatus*) in thicket vegetation: is the cheetah a savanna specialist? *Journal of Zoology* **271:** 310–317.

BLAKE, J.G. & MOSQUERA, D. 2014. Camera trapping on and off trails in lowland forest of eastern Ecuador: Does location matter? *Mastozoología Neotropical* **21(1):** 17-26.

BORAH, J., SHARMA, T., DAS, D., RABHA, N., KAKATI, N., BASUMATARY, A., AHMED, M.F. & VATTAKAVEN, J. 2013. Abundance and density estimates forcommon leopard *Panthera pardus* and clouded leopard *Neofelis nebulosa* in Manas National Park, Assam, India. *Oryx* **48:** 149–155.

BORCHERS, D. 2010. A non-technical overview of spatially explicit capture-recapture models. *Journal of Ornithology* **152:** 435-444.

BORCHERS, D.L. & EFFORD, M.G. 2008. Spatially explicit maximum likelihood methods for capture-recapture studies. *Biometrics* **64:** 377-385.

BÖRGER, L., FRANCONI, N., DE MICHELE, G., GANTZ, A., MESCHI, F., MANICA, A., LOVARI, S. & COULSON, T. 2006. Effects of sampling regime on the mean and variance of home range size estimates. *Journal of Animal Ecology* **75:** 1393-1405.

BOTHMA, J. DU P. & WALKER, C. 1999. *Larger Carnivores of the African Savannas*. Van Schaik Publishers, Pretoria.

BOWLAND, A.E. & MILLS, M.G.L. 1994. The 1990/1991 cheetah photographic survey. *Scientific Report 6/94*. SANParks, Skukuza.

BROOMHALL, L.S., MILLS, M.G.L. & DU TOIT, J.T. 2003. Home range and habitat use by cheetahs (Acinonyx jubatus) in the Kruger National Park. *Journal of Zoology* **261:** 119–128.

BUK, K. & MARNEWICK, K. 2010. Cheetah conservation in South Africa. *African Insight* **39(4):** 212-224.

BULTE, E.H. & RONDEAU, D. 2005. Why compensating for wildlife damages may be bad for conservation. *Journal of Wildlife Management* **69:** 14–19.

BURNHAM, K.P. & ANDERSON, D.R. 2002. *Model selection and multimodel inference: a practical information-theoretic approach.* Second edition. Springer. New York.

BUTLER, J.R.A. 2000. The economic costs of wildlife predation on livestock in Gokwe communal land, Zimbabwe. *African Journal of Ecology* **38:** 23–30.

CARBONE, C., CHRISTIE, S., CONFORTI, K., COULDON, T., FRANKLIN, N., GINSBERG, J.R., GRIFFITHS, M., HOLDEN, J., KAWANISHI, K., KINNAIRD, M., LAIDLAW, R., LYNAM, A., MACDONALD, D.W., MARTYR, D., MCDOUGAL, C., NATH, L., O'BRIEN, T., SEIDENSTICKER, J., SMITH, D.J.L., SUNQUIST, M., TILSON, R. & WAN SHAHRUDDIN, W.N. 2001. The use of photographic rates to estimate densities of tigers and other cryptic mammals. *Animal Conservation* **4:** 75-79.

CARO T.M. 1994. *Cheetahs of the Serengeti Plains: group living in an asocial species.* The University of Chicago Press. Chicago.

CARO, T.M. & COLLINS, D.A. 1987. Male cheetah organization and territoriality. *Ethology* **74(1):** 52-64.

CHATTHA, S.A., IQBAL, S., RASHEED, Z., RAZZAQ, A., HUSAIN, M. & ABBAS, M.N. 2013. Human-leopard conflict in Machiara National Park (MNP), Azad Jamu and Kashmir (AJ and K), Pakistan. *Jgiass* **1:** 17–21.

CLARKE, J. 2012. *Save me from the Lion's Mouth: Exposing Human-Wildlife Conflict in Africa.* Struik Nature. Cape Town.

COPPINGER, R., COPPINGER, L., LANGELOH, G., GETTLER, L. & LORENZ, J. 1988. A decade of use of livestock guarding dogs. *Proceedings of the Thirteenth Vertebrate Pest Conference*, University of Nebraska. Lincoln. **13:** 209–214.

COZZI, G., BROEKHUIS, F., MCNUTT, J.W., SCHMID, B. 2013. Comparison of the effects of artificial and natural barriers on large African carnivores: implications for interspecific relationships and connectivity. *Journal of Animal Ecology* **82:** 707-715.

DALTON, D.L., CHARRUAU, P., BOAST, L. & KOTZE, A. 2013. Social and genetic population structure of free-ranging cheetah in Botswana: implications for conservation. *European Journal of Wildlife Research* **59:** 281–285.

DICKMAN, A.J. 2010. Complexities of conflict: the importance of considering social factors for effectively resolving human-wildlife conflict. *Animal Conservation* **13:** 458–466.

DILLON, A. & KELLY, M.J. 2007. Ocelot *Leopardus pardalis* in Belize: the impact of trap spacing and distance moved on density estimates. *Oryx* **41:** 469–477.

DURANT, S.M. 1998. Competition refuge and coexistence: an example from Serengeti carnivores. *Journal of Animal Ecology* **67:** 370-386.

DURANT, S.M., KELLY, M. & CARO, T.M. 2004. Factors affecting life and death in Serengeti cheetahs: environment, age, and sociality. *Behavioural Ecology* **15(1):** 11- 22.

DURANT, S.M., BASHIR, S., MADDOX, T. & LAURENSON, M.K. 2007. Relating long- term studies to conservation practice: the case of the Serengeti Cheetah Project. *Conservation biology : the journal of the Society for Conservation Biology* **21:** 602–11.

DURANT, S.M., DICKMAN, A.J. MADDOX, T., WAWERU, M.N., CARO, T., PETTORELLI, N., MACDONALD, D.W. & LOVERIDGE, A.J. 2010. Past, present, and future of cheetahs in Tanzania: their behavioural ecology and conservation. In Macdonald, D.W. & Loveridge, A.J. Editors. *Biology and Conservation of Wild Felids.* Oxford University Press. Oxford. P. 373-382.

DWNP (The Department Of Wildlife and National Parks). 2009. National Conservation Action plan for Cheetahs and African Wild Dog in Botswana.

DWNP (The Department Of Wildlife and National Parks). 2012. Aerial census of animals in Botswana: Dry season.

DYER, S. 2012. Population size, demography and spatial ecology of cheetahs in the Timbavati Private Nature Reserve, South Africa. MSc thesis. Rhodes University. Grahamstown.

EATON, R.L. 1970. Group interactions, spacing and territoriality in cheetahs. *Zeitschrift für Tierpsychologie* **27(4):** 482–491.

EFFORD, M. 2004. Density estimation in live-trapping studies. *Oikos* **106:** 598-610.

EFFORD, M.G., DAWSON, D.K. & ROBBINS, C.S. 2004. DENSITY: software for analysing capture – recapture data from passive detector arrays. *Animal Biodiversity and Conservation* **27:** 217–228.

ELLIOT, N.B., VALEIX, M., MACDONALD, D.W. & LOVERIDGE, A.J. 2014. Social relationships affect dispersal timing revealing a delayed infanticide in African lions. *Oikos* **123:** 1049–1056.

ERICSSON, G. & HEBERLEIN, T.A. 2003. Attitudes of hunters, locals, and the general public in Sweden now that the wolves are back. *Biological Conservation* **111:** 149–159.

FERREIRA, S.M.& HOFMEYR, M. 2014. Managing charismatic carnivores in small areas: large felids in South Africa. *South African Journal of Wildlife Research* **44:** 32–42.

FODDY, W.H. 1994. *Constructing Questions for Interviews and Questionnaires: Theory and Practice in Social Research.* Cambridge University Press. Cambridge.

FOSTER, R.J. & HARMSEN, B.J. 2012. A critique of density estimation from camera-trap data. *The Journal of Wildlife Management* **76(2):** 224–236.

FRECKLETON, R. P. 2011. Dealing with collinearity in behavioural and ecological data: model averaging and the problems of measurement error. *Behavioural Ecology and Sociobiology* **65:** 91–101.

GARDNER, B., REPPUCCI, J., LUCHERINI, M. & ROYLE, J.A. 2010. Spatially explicit inference for open populations: estimating demographic parameters from camera-trap studies. *Ecology* **91(11):** 3376-3383.

GERBER, B.D., KARPANTY, S.M.& KELLY, M.J. 2012. Evaluating the potential biases in carnivore capture-recapture studies associated with the use of lures and varying density estimation techniques using photographic-sampling data of the Malagasy civet. *Population Ecology* **54:** 43-54.

GLIEM, J.A. & GLIEM, R.R. 2003. *Calculating, Interpreting, and Reporting Cronbach's Alpha Reliability Coefficient for Likert-Type Scales*. Midwest Research to Practice Conference in Adult, Continuing, and Community Education. pp. 82–88.

GONZALEZ, A., NOVARDO, A., FUNES, M., PAILACURA, O., BOLGERI, M.J. &

WALKER, S. 2012. Mixed-breed guarding dogs reduce conflict between goat herders and native carnivores in Patagonia. *Human-Wildife Interactions* **6:** 327–334.

GOPALASWAMY, A.M., ROYLE, J.A., HINES, J.E., SINGH, P., JATHANNA, D., KUMAR, N.S. & KARANTH, K.U. 2012. Program SPACECAP: software for estimating animal density using spatially explicit capture-recapture models. *Methods in Ecology and Evolution* **3:** 1067-1072.

GOPALASWAMY, A.M., ROYLE, A.J., HINES, J.E., SINGH, P., JATHANNA, D., KUMAR, N.S. & KARANTH, K.U. 2013. Package 'SPACECAP' A program to estimate animal abundance and density using Bayesian spatially explicit capture- recapture models.

GÖTZE, A.R., CILLIERS, S.S., BEZUIDENHOUT, H. & KELLNER, K. 2008. Analysis of the vegetation of the sandstone ridges (ib land type) of the north-eastern parts of the Mapungubwe National Park, Limpopo Province, South Africa. *Koedoe: African Protected Area Conservation and Science* **50:** 72–81.

GRAHAM, K., BECKERMAN, A.P. & THIRGOOD, S. 2005. Human–predator–prey conflicts: ecological correlates, prey losses and patterns of management. *Biological Conservation* **122:** 159–171.

GRANT, T. 2012. Leopard population density, home range size and movement patterns in a mixed landuse area of the Mangwe district of Zimbabwe. MSc thesis, Rhodes University. Grahamstown.

GRAY, T.N.E. & PRUM, S. 2012. Leopard density in post-conflict landscape, Cambodia: Evidence from spatially explicit capture-recapture. *The Journal of Wildlife Management* **76(1):** 163–169.

GROS, P.M. 1998. Status of the cheetah *Acinonyx jubatus* in Kenya: a field-interview assessment. *Biological Conservation* **85:** 137-149.

GROS, P.M. 2000. Status of the cheetah in Tanzania in the mid 1990's. *Journal of East African Natural History* **89:** 85–100.

GROS, P.M. 2002. The status and conservation of the cheetah *Acinonyx jubatus* in Tanzania. *Biological Conservation* **106:** 177–185.

GROS, P.M., KELLY, M. J. & CARO, T. M. 1996. Estimating carnivore densities for conservation purposes: indirect methods compared to baseline demographic data. *Oikos* **77:** 197–206.

HALL-MARTIN, A.J. & BOTHA, B.P. 1980. A note on feeding habits, ectoparasites and measurements of black-backed jackal *Canis mesomelas* from Addo Elephant National Park. *Koedoe* **23:** 157-162.

HARRIS, S., CRESSWELL, W.J., FORDE, P.G., TREWHELLA, W.J., WOOLLARD, T. &

WRAY, S. 1990. Home-range analysis using radio-tracking data: a review of the problems and techniques particularly as applied to the study of mammals. *Mammal Review* **20:** 97-123.

HAYWARD, M.W. & SLOTOW, R. 2009. Temporal partitioning of activity in large African carnivores: tests of multiple hypotheses. *South African Journal of Wildlife Research* **39(2):** 109-125.

HEBERLEIN, T.A. 2012. *Navigating Environmental Attitudes*. Oxford University Press. New York.

HEILBRUN, R.D., SILVY, N.J., TEWES, M.E. & PETERSON, M.J. 2003. Using automatically triggered cameras to individually identify bobcats. *Wildlife Society Bulletin* **31:** 748–755.

HENSCHEL, P. & RAY, J. 2003. Leopards in African Rainforests: Survey and monitoring techniques. *WCS Global Carnivore Program.*

HOLMERN, T., NYAHONGO, J. & RØSKAFT, E. 2007. Livestock loss caused by predators outside the Serengeti National Park, Tanzania. *Biological Conservation* **135:** 518–526.

HOLMERN, T. & RØSKAFT, E. 2013. The poultry thief: Subsistence farmers' perceptions of depredation outside the Serengeti National Park, Tanzania. *African Journal of Ecology* **52:** 3334–342.

HOLMERN, T., NYAHONGO, J. & RØSKAFT, E. 2007. Livestock loss caused by predators outside the Serengeti National Park, Tanzania. *Biological Conservation* **135(4):** 518– 526.

HOUSER, A., SOMERS, M. J. & BOAST, L.K. 2009. Home range use of free-ranging cheetah on farm and conservation land in Botswana. *South African Journal of Wildlife Research* **39(1):** 11-22.

HUDENKO, H.W. 2012. Exploring the influence of emotion on human decision making in human–wildlife conflict. *Human Dimensions of Wildlife: An International Journal* **17:** 16–28.

HUNTER, L. 1998. Early post-release movements and behaviour of reintroduced cheetahs and lions, and technical considerations in large carnivore restoration. *Proceedings of a Symposium on Cheetahs as Game Ranch Animals.* Ondersterpoort: 72-82.

IUCN (International Union for Conservation of Nature). 2013. *IUCN Red List of Threatened Species*. Version 2013.2. Available at: http://www.iucnredlist.org. Accessed: 10 April 2014.

JACKSON, T.P., MOSOJANE, S., FERREIRA, S.M. & VAN AARDE, R.J. 2008. Solutions for elephant *Loxodonta africana* crop raiding in northern Botswana: moving away from symptomatic approaches. *Oryx* **42:** 83–91.

JACKSON, R.M., ROE, J.D., WANGCHUK, R. & HUNTER, D.O. 2010. Estimating snow leopard population abundance using photography and capture-recapture techniques. *Wildlife Society Bulletin* **34(3):** 772-781.

JACKSON, C.R., MCNUTT, J.W. & APPS, P.J. 2012. Managing the ranging behaviour of African wild dogs *(Lycaon pictus*) using translocated scent marks. *Wildlife Research* **39:** 31–34.

JENNESS, J. 2012. Repeating shapes for ArcGIS. Jenness Enterprises. Available at: http://www.jennessent.com/arcgis/repeat_shapes.htm. Accessed: 12 January 2014.

JOUBERT, S.C.J. 1984. Naledi Game Reserve: Report on an ecological survey of the Eastern Tuli Game Reserve. Unpublished report.

KALLE, R., RAMESH, T., QURESHI, Q. & SANKAR, K. 2011. Density of tiger and leopard in a tropical deciduous forest of Mudumalai Tiger Reserve, southern India, as estimated using photographic capture–recapture sampling. *Acta Theriologica* **56:** 335– 342.

KARANTH, K.U. 1995. Estimating tiger *Panthera tigris* populations from camera-trap data using capture-recapture models. *Biological Conservation* **71:** 333-338.

KARANTH, K.U. & NICHOLS, J.D. 1998. Estimation of tiger densities in India using photographic captures and recaptures. *Ecology* **79(8):** 2852-2862.

KARANTH, K.U. & NICHOLS, J.D. 2002. *Monitoring tigers and their prey: a manual for researchers, managers and conservationists in tropical Asia*. Centre for Wildlife Studies. Bangalore, India.

KARANTH, K.U., FUNSTON, P. & SANDERSON, E. MACDONALD, D.W. & LOVERIDGE, A.J. 2010. Many ways of skinning a cat: tools and techniques for studying wild felids. In Macdonald, D.W. & Loveridge, A.J. Editors. *Biology and Conservation of Wild Felids.* Oxford University Press. Oxford. p. 197-216.

KELLY, M.J. 2001. Computer-aided photography matching in studies using individual identification: an example from Serengeti cheetahs. *Journal of Mammalogy* **82(2):** 440- 449.

KELLY, M. J. & DURANT, S. M. 2000. Viability of the Serengeti Cheetah Population. *Conservation Biology* **14:** 786–797.

KELLY, M.J., LAURENSON, M.K., FITZGIBBON, C.D., COLLINS, D.A., DURANT, S.M., FRAME, G.W., BERTRAM, B.C.R. & CARO, T.M. 1998. Demography of the Serengeti cheetah (*Acinonyx jubatus*) population: the first 25 years. *Journal of Zoology* **244:** 473-488.

KELLY, M.J., NOSS, A.J., DI BITETTI, M.S., MAFFEI, L., ARISPE, R.L., PAVIOLO, A., DE ANGELO, C.D. & DI BLANCO Y.E. 2008. Estimating Puma densities from camera trapping across three study sites: Bolivia, Argentina, and Belize. *Journal of Mammalogy* **89(2):** 408-418.

KEMP, L.V. & MILLS, M.G.L. 2005. The 4th wild dog and 2nd cheetah photographic census in the greater Kruger region, September 2004 – April 2005. Unpublished report. Endangered Wildlife Trust, Johannesburg.

KGATHI, D.L., MMOPELWA, G., MASHABE, B. & MOSEPELE, K. 2012. Livestock predation, household adaptation and compensation policy: a case study of Shorobe Village in

northern Botswana. *Agrekon: Agricultural Economics Research, Policy and Practice in Southern Africa* **51:** 22–37.
KLEIN, R. 2007. Status Report for the Cheetah in Botswana. Published by Cheetah Conservation Botswana for the IUCN's Southern Africa Regional Status Report. 14–21.
KLEIN, R. 2013. An assessment of human carnivore conflict in the Kalahari region of Botswana. MSc thesis. Rhodes University. Grahamstown.
KNIGHT, A. 1999. Cheetah numbers in a changing environment: Kalahari Gemsbok National Park. Endangered Wildlife Trust, Johannesburg.
LANDA, A., GUDVANGEN, K., SWENSON, J E. & RØSKAFT, E. 1999. Factors associated with wolverine *Gulo gulo* predation on domestic sheep. *Journal of Animal Ecology* **36:** 963–973.
LAURENSON, M.K. 1994. High juvenile mortality in cheetahs (*Acinonyx jubatus*) and its consequences for maternal care. *Journal of Zoology* **234:** 387-408.
LAURENSON, M.K., WIELEBNOWSKI, N. & CARO, T.M. 1995. Extrinsic factors and juvenile mortality in cheetahs. *Conservation Biology* **9:** 1329–1331.
LIND, P. 1974. Shashe Limpopo Ranger Report 1973/10974. *Unpublished report.*
LINDSEY, P., DU TOIT, J.T. & MILLS, M.G.L. 2004. The distribution and population status of African wild dogs (*Lycaon pictus*) outside protected areas in South Africa. *South African Journal of Wildlife Research* **34(2):** 143-151.
LINDSEY, P.A., HAVEMANN, C.P., LINES, R., PALAZY, L., PRICE, A.E., RETIEF, T.A., RHEBERGEN, T. & VAN DER WAAL, C. 2013. Determinants of persistence and tolerance of carnivores on Namibian ranches: implications for conservation on southern African private lands. *PLoS ONE* **8(1):** e52458.
LINNELL, J.D.C., SWENSON, J.E. & ANDERSEN, R. 2001. Predators and people: conservation of large carnivores is possible at high human densities if management policy is favourable. *Animal Conservation* **4:** 345–349.
LONG, R.A., MACKAY, P., ZIELINSKI, W.J. & RAY, J.C. 2008. *Noninvasive Survey Methods for Carnivores*. Island Press, Washington.
MACDONALD, D.W., LOVERIDGE, A.J. & NOWELL, K. 2010a. Dramatis personae: an introduction to the wild felids. In Macdonald, D.W. & Loveridge, A.J. Editors. *Biology and Conservation of Wild Felids.* Oxford University Press. Oxford. p. 3-58.
MACDONALD, D.W., LOVERIDGE, A.J. & RABINOWITZ, A. 2010b. Felid futures: crossing disciplines, borders, and generations. In Macdonald, D.W. & Loveridge, A.J. Editors. *Biology and Conservation of Wild Felids.* Oxford University Press. Oxford. p. 599-649.

MADDOCK, A.H. & MILLS, M.G.L. 1994. Population characteristics of African wild dogs *lycaon pictus* in the Eastern Transvaal Lowveld, South Africa, as revealed through photographic records. *Biological Conservation* **67:** 57-62.

MAFFEI, L. & NOSS, A.J. 2008. How small is too small? Camera trap survey areas and density estimates for ocelots in the Bolivian Chaco. *Biotropica* **40(1)**: 71–75.

MAFFEI, L., NOSS, A.J., CUELLAR, E. & RUMIZ, D.I. 2005. Ocelot (Felis pardalis) population densities, activity, and ranging behaviour in the dry forests of eastern Bolivia: data from camera trapping. *Journal of Tropical Ecology* **21:** 349-353.

MARKER, L. 1998. The current status of the cheetah (Acinonyx jubatus). Preceedings of a Symposium on Cheetahs as Game Ranch Animals, Onderstepoort.

MARKER, L. 2000. Aspects of the ecology of the cheetah (*Acinonyx jubatus*) on the north central Namibian farmlands. *Namibian Scientific Society Journal* **48:** 40-48.

MARKER, L. L. 2002. Aspects of cheetah (Acinonyx jubatus) biology, ecology and conservation strategies on Namibian Farmlands. PhD thesis. University of Oxford. Oxford.

MARKER, L.L., MILLS, M.G.L. & MACDONALD, D.W. 2003a. Factors influencing perceptions of conflict and tolerance toward cheetahs on namibian farmlands. *Conservation Biology* **17:** 1290–1298.

MARKER, L.L., DICKMAN, A.J., JEO, R.M., MILLS, M.G.L. & MACDONALD, D.W. 2003b. Demography of the Namibian cheetah, Acinonyx jubatus jubatus. *Biological Conservation* **114:** 413–425.

MARKER, L.L., DICKMAN, A.J., MILLS, M.G.L. & MACDONALD, D.W. 2003c. Aspects of the management of cheetahs, *Acinonyx jubatus jubatus*, trapped on Namibian farmlands. *Biological Conservation* **114:** 401–412.

MARKER, L.L., MUNTIFERING, J.R., DICKMAN, A.J., MILLS, M.G.L. & MACDONALD, D. W. 2003d. Quantifying prey preferences of free-ranging Namibian cheetahs. *South African Journal of Wildlife Research* **33:** 43–53.

MARKER, L., DICKMAN, A. & SCHUMANN, M. 2005. Using livestock guarding dogs as a conflict resolution strategy on Namibian farms. *Carnivore Damage Prevention News*: 28–32.

MARKER, L., DICKMAN, A., WILKINSON, C., SCHUMANN, B. & FABIANO, E. 2007. The Namibian Cheetah : Status Report. *CAT News*: 4–13.

MARKER, L.L., DICKMAN, A.J., MILLS, M.G.L., JEO, R.M. & MACDONALD, D.W. 2008a. Spatial ecology of cheetahs on north-central Namibian farmlands. *Journal of Zoology* **274:** 226-238.

MARKER, L.L., FABIANO, E. & NGHIKEMBUA, M. 2008b. The use of remote camera traps to estimate density of free-ranging cheetahs in North-Central Namibia. CAT *News* **49:** 22-24.

MARKER, L., DICKMAN, A.J., MILLS, M.G.L. MACDONALD, D.W. & LOVERIDGE, A. 2010. Cheetahs and ranchers in Namibia: a case study. In Macdonald, D.W. & Loveridge, A.J. Editors. *Biology and Conservation of Wild Felids.* Oxford University Press. Oxford. P. 353-372.

MARNEWICK, K. & CILLIERS, D. 2006. Range use of two coalitions of male cheetahs, *Acinonyx jubatus,* in the Thabazimbi district of the Limpopo Province, South Africa. *South African Journal of Wildlife Research* **36(2):** 147–151.

MARNEWICK, K. & DAVIES-MOSTERT, H.T. 2012. *Kruger National Park 2008/2009 5th Wild Dog and 2nd Cheetah Photographic Census*. Report to the Endangered Wildlife Trust and SANParks

MARNEWICK, K.A., BOTHMA, J. du P. & VERDOORN, G.H. 2006. Using camera- trapping to investigate the use of a tree as a scent-marking post by cheetahs in the Thabazimbi district. *South African Journal of Wildlife Research* **36(2):** 139-145.

MARNEWICK, K., BECKHELLING, A., CILLIERS, D., LANE, E., MILLS, G., HERING, K., CALDWELL, P., HALL, R. & MEINTJES, S. 2007. The Status of the Cheetah in South Africa. *CAT News*: 22–31.

MARNEWICK, K., FUNSTON, P.J. & KARANTH, K.U. 2008. Evaluating camera trapping as a method for estimating cheetah abundance in ranching areas. *South African Journal of Wildlife Research* **38(1):** 59-65.

MARNEWICK, K., FERREIRA, S.M., GRANGE, S., WATERMEYER, J., MAPUTLA, N. & DAVIES-MOSTERT, H.T. 2014. Evaluating the status of and African wild dogs *Lycaon pictus* and cheetahs *Acinonyx jubatus* through tourist-based photographic surveys in the Kruger National Park. *PloS one* **9:** 1–8.

MCKENZIE, A.A. 1990. Co-operative hunting in the black-backed jackal *Canis mesomelas* Schreber. PhD thesis. University of Pretoria, Pretoria.

MILLS, M.G.L. 1989. Cheetah and wild dog research in the Kruger National Park in 1988. A progress report. *Quagga*: 5-6.

MISHRA, C., ALLEN, P., MCCARTHY, T., MADHUSUDAN, M.D., BAYARJARGAL, A. & PRINS, H.H.T. 2003. The role of incentive programs in conserving the snow leopard. *Conservation Biology* **17:** 1512–1520.

MONDAL, K., GUPTA, S., BHATTACHARJEE, S., QURESHI, Q. & SANKAR, K. 2012. Response of leopards to re-introduced tigers in Sariska Tiger Reserve, Western India. *International Journal of Biodiversity and Conservation* **4(5):** 228–236.

MYERS, N. 1975. The cheetah, *Acinonyx jubatus*, in Africa. IUCN Monograph No. 4, International Union for Nature and Natural Resources, Morges, Switzerland.

NCHUNGA, M.L. 1978. A study of the potential for the commercial use of wildlife in the North-eastern Tuli Block. MSc thesis. Wildlife and Fisheries Sciences, Graduate College of Texas A & M University.

NEGRÕES, N., SARMENTO, P., CRUZ, J., EIRA, C., REVILLA, E., FONSECA, C., SOLLMANN, R., TÕRRES, N.M., FURTADO, M.M., JÁCOMO, A.T.A. & SILVEIRA, L. 2010. Use of camera-trapping to estimate puma density and influencing factors in central Brazil. *Wildlife Management* **74(6):** 1195-1203.

NEGRÕES, N., SOLLMANN, R., FONSECA, C., JÁCOMO, A.T.A., REVILLA, E. & SILVEIRA, L. 2012. One or two cameras per station? Monitoring jaguars and other mammals in the Amazon. *Ecological Research* **27:** 639-648.

NICHOLS, J.D. 1992. Capture-recapture models. *Biological Science* **42(2):** 94-101. NOSS, A.J., GARDNER, B., MAFFEI, L., CUÉLLAR, E., MONTAÑO, R., ROMERO-MUNOZ, A., SOLLMAN, R. & O'CONNELL, A.F. 2012. Comparison of density estimation methods for mammal populations with camera traps in the Kaa-Iya del Gran Chaco landscape. *Animal Conservation* **15:** 527–535.

NOSS, A., POLISAR, J., MAFFEI, L., GARCIA, R. & SILVER, S. 2013. Evaluating jaguar densities with camera traps. Jaguar Conservation Program, *Wildlife Conservation Society*. New York.

O'BRIEN, T.G. & KINNAIRD, M.F. 2011. Density estimation of sympatric carnivores using spatially explicit capture-recapture methods and standard trapping grid. *Ecological Applications* **21(8):** 2908-2916.

O'BRIEN, T.G., KINNAIRD, M.F. & WIBISONO, H.T. 2003. Crouching tigers, hidden prey: Sumatran tiger and prey populations in a tropical forest landscape. *Animal Conservation* **6:** 131 139.

O'CONNELL, A.F., NICHOLS, J.D. & KARANTH, K.U. 2011. *Camera Traps in Animal Ecology Methods and Analyses.* Springer, Tokyo.

OGADA, M.O., WOODROFFE, R., OGUGE, N.O. & FRANK, L.G. 2003. Limiting Depredation by African Carnivores: the Role of Livestock Husbandry. *Conservation Biology* **17(6):** 1521–1530.

OGARA, W.O., GITAHI, N.J., ANDANJE, S.A., OGUGE, N., NDUATI, D.W., & MAINGA, A.O. 2010. Determination of carnivores prey base by scat analysis in Samburu community group ranches in Kenya. *African Journal of Environmental Science and Technology* **4(8):** 540–546.

OLSON, K. 2010. An examination of questionnaire evaluation by expert reviewers.

Sociology Department, Faculty Publications. Paper 136. http://digitalcommons.unl.edu/sociologyfacpub/136. Accessed: 1 October 2014.

OTIS, D.L., BURNHAM, K.P., WHITE, G.C. & ANDERSON, D.R. 1978. Statistical inference from capture data on closed animal populations. *Wildlife Monographs* **62:** 3- 135.

PACKER, C., HOLT, R.D., HUDSON, P.J., LAFFERTY, K.D. & DOBSON, A.P. 2003. Keeping the herds healthy and alert: implications of predator control for infectious disease. *Ecology Letters* **6:** 797–802.

PARKER, D.M., WHITTINGTON-JONES, B.M., BERNARD, R.T.F. & DAVIES- MOSTERT, H.T. 2014. Attitudes of rural communities toward dispersing african wild dogs in South Africa. *Human Dimensions of Wildlife* **19:** 512–522.

PATTERSON, B.D., KASIKI, S.M., SELEMPO, E. & KAYS, R.W. 2004. Livestock predation by lions (*Panthera leo*) and other carnivores on ranches neighboring Tsavo National Park, Kenya. *Biological Conservation* **119:** 507–516.

PETTIFER, H.L. 1981. Aspects on the ecology of cheetahs (*Acinonyx jubatus*) on the Suikerbosrand Nature Reserve. In: *Proceedings of the First World Furbearer Conference.* (eds) J.A. Chapman & D. Punsley, pp. 1121-1142. University of Maryland, Frostburg. Virginia.

PETTORELLI, N., HILBORN, A., BROEKHUIS, F. & DURANT, S.M. 2009. Exploring habitat use by cheetahs using ecological niche factor analysis. *Journal of Zoology* **277:** 141–148.

POLISAR, J., MAXIT, I., SCOGNAMILLO, D., FARRELL, L., SUNQUIST, M.E. & EISENBERG, J.F. 2003. Jaguars, pumas, their prey base, and cattle ranching: ecological interpretations of a management problem. *Biological Conservation* **109:** 297–310.

PRUGH, L R., STONER, C.J., EPPS, C.W., BEAN, W.T., RIPPLE, W.J., LALIBERTE, A.S. & BRASHARES, J.S. 2009. The rise of the mesopredator. *BioScience* **59:** 779– 791.

PURCHASE, G.K. & DU TOIT, J.T. 2000. The use of space and prey by cheetah in Matusadona National Park, Zimbabwe. *South African Journal of Wildlife Research* **30(4):** 139-144.

PURCHASE, G., MARKER, L., MARNEWICK, K., KLEIN, R. & WILLIAMS, S. 2007. Regional Assessment of the Status, Distribution and Conservation Needs of Cheetahs in Southern Africa. *CAT News*: 44–46.

RAMAKRISHNAN, U., COSS, R.G. & PELKEY, N.W. 1999. Tiger decline caused by the reduction of large ungulate prey: evidence from a study of leopard diets in southern India. *Biological Conservation* **89:** 113–120.

RAY, J.C., HUNTER, L. & ZIGOURIS, J. 2005. Setting conservation and research priorities for larger African carnivores. Wildlife Conservation Society, New York, USA.

RØSKAFT, E., HÄNDEL, B., BJERKE, T. & KALTENBORN, B.P. 2007. Human attitudes towards large carnivores in Norway. *Wildlife Biology* **13:** 172–185.

ROWE, R.J. 2009. Environmental and geometric drivers of small mammal diversity along elevational gradients in Utah. *Ecography* **32:** 411–422.

ROYLE, J.D., KARANTH, K.U., GOPALASWAMY, A.M. & KUMAR, N.S. 2009a. Bayesian inference in camera trapping studies for a class of special capture-recapture models. *Ecological Society of America* **90(11):** 3233-3244.

ROYLE, J.D., NICHOLS, J.D., KARANTH, K.U. & GOPALASWAMY, A.M. 2009b. A hierarchical model for estimating density in camera-trap studies. *Journal of Applied Ecology* **46:** 118-127.

SANTOS, J.R.A. 1999. Cronbach's Alpha: a tool for assessing the reliability of scales. *Extension Information Technology* **37(2):** 1-4.

SCHIESS-MEIER, M., RAMSAUER, S., GABANAPELO, T. & KÖNIG, B. 2007. Livestock predation—insights from problem animal control registers in Botswana. *Journal of Wildlife Management* **71:** 1267–1274.

SEAMAN, D E. & POWELL, R.A. 1996. An evaluation of the accuracy of kernel density estimators for home range analysis. *Ecology* **77:** 2075–2085.

SELEBATSO, M., MOE, S.R. & SWENSON, J.E. 2008. Do farmers support cheetah *Acinonyx jubatus* conservation in Botswana despite livestock depredation? *Oryx* **42(3):** 430-436.

SELIER, S. J. 2007. The social structure, distribution and demographic status of the African elephant population in the Central Limpopo River Valley of Botswana, Zimbabwe and South Africa. MSc thesis. University of Pretoria. Pretoria.

SIFUNA, N. 2010. Wildlife damage and its impact on public attitudes towards conservation: a comparative study of Kenya and Botswana, with particular reference to Kenya's Laikipia Region and Botswana's Okavango Delta Region. *Journal of Asian and African Studies* **45:** 274–296.

SILVER, S. 2004. Assessing jaguar abundance using remotely triggered cameras. Jaguar Conservation Program. *Wildlife Conservation Society.*

SILVERTOWN, J. 2009. A new dawn for citizen science. *Trends in ecology & evolution* **24:** 467–471

SKINNER, J.D. & CHIMIMBA, C.T. 2005. *The Mammals of the Southern African Subregion.* Cambridge University Press, Cambridge.

SNYMAN, A., JACKSON, C.R. & FUNSTON, P.J. 2014. The effect of alternative forms of hunting on the social organization of two small populations of lions *Panthera leo* in southern Africa. *Oryx*: 1–7.

SOISALO, M.K. & CAVALCANTI, S.M.C. 2006. Estimating the density of a jaguar population in the Brazilian Pantanal using camera-traps and capture-recapture sampling in combination with GPS radio-telemetry. *Biological Conservation* **129:** 487-496.

SOLLMANN, R., FURTADO, M.M., GARDNER, B., HOFER, H., JÁCOMO, A.T.A., TÔRRES, N.M. & SILVEIRA, L. 2011. Improving density estimates for elusive carnivores: accounting for sex-specific detection and movements using spatial capture– recapture models for jaguars in central Brazil. *Biological Conservation* **144:** 1017–1024.

SOSO, S.B., KOZIEL, J.A., JOHNSON, A., LEE, Y.J. & FAIRBANKS, W.S. 2014. Analytical methods for chemical and sensory characterization of scent-marking in large wild mammals: a review. *Sensors* **14:** 4428-4465.

STANLEY, T.R. & BURNHAM, K.P. 1999. A closure test for time–specific capture- recapture data. *Environmental and Ecological Statistics* **6:** 197-209.

STEYN, T. 2004. *Northern Tuli Game Reserve: Memories of the Founding of a Major Private Game Reserve.* E & R Steyn. Kelvin, Gauteng.

SUN, C.C., FULLER, A.K. & ROYLE, J.A. 2014. Trap configuration and spacing influences parameter estimates in spatial capture-recapture models. *PloS ONE* **9(2):** 1-9.

THOMPSON, S.K., WHITE, G.C. & GOWAN, C. 1998. *Monitoring vertebrate populations.* Academic Press, New York, United States of America.

THORN, M., GREEN, M., DALERUM, F., BATEMAN, P.W. & SCOTT, D.M. 2012. What drives human – carnivore conflict in the North West Province of South Africa ? *Biological Conservation* **150:** 23–32.

THORN, M., GREEN, M., MARNEWICK, K. & SCOTT, D.M. 2014. Determinants of attitudes to carnivores: implications for mitigating human–carnivore conflict on South African farmland. *Oryx*: 1–8.

TOBLER, M.W. & POWELL, G.V N. 2013. Estimating jaguar densities with camera traps: problems with current designs and recommendations for future studies. *Biological Conservation* **159:** 109–118.

TOBLER, M.W., CARRILLO-PERCASTEGUI, S.E., LEITE PITMAN, R., MARES, R. & POWELL, G. 2008. An evaluation of camera traps for inventorying large- and medium- sized terrestrial rainforest mammals. *Animal Conservation* **11:** 169–178.

TOBLER, M.W., CARRILLO-PERCASTEGUI, S.E., ZÚÑIGA HARTLEY, A. & POWELL, G.V.N. 2013. High jaguar densities and large population sizes in the core habitat of the southwestern Amazon. *Biological Conservation* **159:** 375–381.

TROLLE, M. & KERY, M. 2003. Estimation of ocelot density in the Pantanal using capture-recapture analysis of camera-trapping data. *Journal of Mammalogy,* **84(2):** 607-614.

WALKER, C.H. 1971. Field impressions: Tuli Circle Rhodesia, Northern Tuli Block Botswana. Unpublished Report.

WALPOLE, M.J. & GOODWIN, H.J. 2001. Local attitudes towards conservation and tourism around Komodo National Park, Indonesia. *Environmental Conservation* **28:** 160–166.

WATERMEYER, J. 2012. Anthropogenic threats to resident and dispersing African wild dogs west and south of the Kruger National Park, South Africa. MSc. thesis. Rhodes University. Grahamstown.

WHITE, G.C., ANDERSON, D.R., BURNHAM, K.P. & OTIS, D.L. 1982. *Capture- recapture and removal methods for sampling closed populations.* Los Alamos National Laboratory Publications LA-8787-NERP. Los Alamos, New Mexico, United States of America.

WHITE, P.C.L., JENNINGS, N.V., RENWICK, A.R. & BARKER, N.H.L. 2005. Questionnaires in ecology: a review of past use and recommendations for best practice. *Journal of Applied Ecology* **42:** 421–430.

WINTERBACH, H.E.K., WINTERBACH, C.W. & SOMERS, M.J. 2014. Landscape suitability in Botswana for the conservation of its six large African carnivores. *PloS one* **9:** 1–12.

WOODROFFE, R. 2000. Predators and people: using human densities to interpret declines of large carnivores. *Animal Conservation* **3:** 165–173.

WOODROFFE, R. & GINSBERG, J.R. 1998. Edge Effects and the Extinction of Populations Inside Protected Areas. *Science* **280:** 2126–2128.

WOODROFFE, R. & GINSBERG, J.R. 1999. Conserving the African wild dog *Lycaon pictus*. I. Diagnosing and treating causes of decline. *Oryx* **33:** 132–142.

WORLD BANK 1983. *Botswana Livestock Subsector Memorandum: Eastern Africa Projects Department*, Southern Agriculture Division, Botswana. Report No. 4665-BT.

WORTON, B.J. 1987. A review of models of home range for animal movement. *Ecological Modelling* **38:** 277-298.

WORTON, B.J. 1995. Using Monte Carlo simulation to evaluate kernel-based home range estimators. *Journal of Wildlife Management* **59(4):** 794-800.

YIRGA, G., IMAM, E., DE IONGH, H.H., LEIRS, H., KIROS, S., YOHANNES, T.G., TEFERI, M. & BAUER, H. 2014. Local spotted hyena abundance and community tolerance of depredation in human-dominated landscapes in Northern Ethiopia. *Mammalian Biology* **79:** 325–330.

APPENDICES

APPENDIX I: List of the common large and medium-sized mammal species in the Northern Tuli Game Reserve, Botswana

ORDER TUBULIDENTATA	
Aardvark	*Orycteropus afer*
ORDER HYRACOIDEA	
Rock hyrax	*Procavia capensis*
ORDER PROBOSCIDEA	
African elephant	*Loxodonta africana*
ORDER LAGOMORPHA	
Scrub hare	*Lepus saxatilis*
ORDER RODENTIA	
Porcupine	*Hysterix africaeaustralis*
Springhare	*Pedetes capensis*
ORDER PRIMATE	
Chacma baboon	*Papio ursinus*
Vervet monkey	*Chlorocebus pygerythrus*
ORDER CARNIVORA	
Aardwolf	*Proteles cristatus*
African civet	*Civettictis civetta*
African clawless otter	*Anonyx capensis*
African wild cat	*Felis lybica*
African wild dog	*Lycaon pictus*
Banded Mongoose	*Mungos mungo*
Bat-eared fox	*Otocyon megalotis*
Black-backed jackal	*Canis mesomelas*
Brown hyena	*Hyaena brunnea*
Cheetah	*Acinonyx jubatus*
Honey badger	*Mellivora capensis*
Large-spotted genet	*Genetta tigrina*
Leopard	*Panthera pardus*
Lion	*Panthera leo*
Selous' mongoose	*Paracynictis selousi*
Slender mongoose	*Galerella sanguinea*
Small-spotted genet	*Genetta genetta*
Spotted hyena	*Crocuta crocuta*
Striped polecat	*Ictonyx striatus*
ORDER PERISSODACTYLA	
Burchell's Zebra	*Equus burchellii*
ORDER SUIFORMES	
Bush pig	*Potomachoerus porcus*
Warthog	*Phacochoerus africanus*
ORDER WHIPPOMORPHA	
Hippopotamus	*Hippopotamus amphibius*
ORDER RUMINANTIA	
Blue Wildebeest	*Connochaetes taurinus*

Bushbuck	*Tragelaphus scriptus*
Common Duiker	*Sylvicapra grimmia*
Eland	*Tragelaphus oryx*
Giraffe	*Giraffa camelopardalis*
Impala	*Aepyceros melampus*
Klipspringer	*Oreotragus oreotragus*
Kudu	*Tragelaphus strepsiceros*
Steenbok	*Raphicerus campestris*
Waterbuck	*Kobus ellipsiprymnus*

Appendix II: Photographic recording rate of mammal species during the first camera trapping survey in the Northern Tuli Game Reserve. The index of relative abundance (RAI) is calculated as the average number of captures per 100 trapping occasions. Species percentage (Spp. %) is the number of capture events (*n*) to the total number of animal photographs.

Species		***n***	**Spp. %**	**RAI**
Aardvaark	*Orycteropus afer*	8	0.24%	0.08
Aardwoolf	*Proteles cristatus*	3	0.09%	0.03
African Civet	*Civettictis civetta*	2	0.06%	0.02
African Elephant	*Loxodonta africana*	473	14.14%	4.73
African wildcat	*Felis lybica*	5	0.15%	0.05
Banded mongoose	*Mungos mungo*	1	0.03%	0.01
Black-backed jackal	*Canis mesomelas*	74	2.21%	0.74
Blue wildebeest	*Connochaetes taurinus*	71	2.12%	0.71
Burchell’s Zebra	*Equus burchellii*	175	5.23%	1.75
Bushbuck	*Tragelaphus scriptus*	1	0.03%	0.01
Bushpig	*Potomachoerus porcus*	1	0.03%	0.01
Chacma Baboon	*Papio ursinus*	133	3.97%	1.33
Cheetah	*Acinonyx jubatus*	9	0.27%	0.09
Common duiker	*Sylvicapra grimmia*	9	0.27%	0.09
Eland	*Tragelaphus oryx*	211	6.31%	2.11
Giraffe	*Giraffa camelopardalis*	250	7.47%	2.5
Honey badger	*Mellivora capensis*	2	0.06%	0.02
Impala	*Aepyceros melampus*	1309	39.12%	13.09
Kudu	*Tragelaphus strepsiceros*	40	1.20%	0.4
Large-spotted genet	*Genetta tigrina*	5	0.15%	0.05
Leopard	*Panthera pardus*	17	0.51%	0.17
Lion	*Panthera leo*	1	0.03%	0.01
Porcupine	*Hysterix africaeaustralis*	6	0.18%	0.06
Scrub hare	*Lepus saxatilis*	64	1.91%	0.64
Selous' mongoose	*Paracynictis selousi*	3	0.09%	0.03
Small-spotted genet	*Genetta genetta*	2	0.06%	0.02
Spotted hyena	*Crocuta crocuta*	120	3.59%	1.2
Steenbok	*Raphicerus campestris*	39	1.17%	0.39

Tree squirrel	*Paraxerus cepapi*	5	0.15%	0.05
Vervet monkey	*Chlorocebus pygerythrus*	13	0.39%	0.13
Warthog	*Phacochoerus africanus*	88	2.63%	0.88
Waterbuck	*Kobus ellipsiprymnus*	2	0.06%	0.02

Appendix III: Photographic recording rate of mammal species during the second camera trapping survey in the Northern Tuli Game Reserve. The index of relative abundance (RAI) is calculated as the average number of captures per 100 trapping occasions. Species percentage (Spp. %) is the number of capture events (*n*) to the total number of animal photographs.

Species		*n*	Spp. %	RAI
Aardvaark	*Orycteropus afer*	3	0.09%	0.03
Aardwoolf	*Proteles cristatus*	2	0.06%	0.02
African Civet	*Civettictis civetta*	4	0.12%	0.04
Elephant	*Loxodonta africana*	56	1.69%	0.56
African wildcat	*Felis lybica*	1	0.03%	0.01
Banded mongoose	*Mungos mungo*	0	0.00%	0
Bat	Species unidentifiable	2	0.06%	0.02
Black-backed jackal	*Canis mesomelas*	73	2.20%	0.73
Blue wildebeest	*Connochaetes taurinus*	88	2.65%	0.88
Burchell’s Zebra	*Equus burchellii*	90	2.71%	0.9
Bushbuck	*Tragelaphus scriptus*	0	0.00%	0
Bushpig	*Potomachoerus porcus*	1	0.03%	0.01
Chacma Baboon	*Papio ursinus*	73	2.20%	0.73
Cheetah	*Acinonyx jubatus*	53	1.59%	0.53
Common duiker	*Sylvicapra grimmia*	3	0.09%	0.03
Eland	*Tragelaphus oryx*	99	2.98%	0.99
Giraffe	*Giraffa camelopardalis*	336	10.11%	3.36
Honey badger	*Mellivora capensis*	1	0.03%	0.01
Impala	*Aepyceros melampus*	213	6.41%	2.13
Kudu	*Tragelaphus strepsiceros*	96	2.89%	0.96
Large-spotted genet	*Genetta tigrina*	0	0.00%	0
Leopard	*Panthera pardus*	13	0.39%	0.13
Lion	*Panthera leo*	5	0.15%	0.05
Mouse	Species unidentifiable	3	0.09%	0.03
Porcupine	*Hysterix africaeaustralis*	8	0.24%	0.08
Scrub hare	*Lepus saxatilis*	23	0.69%	0.23
Selous' mongoose	*Paracynictis selousi*	0	0.00%	0
Small-spotted genet	*Genetta genetta*	10	0.30%	0.1

Spotted hyena	*Crocuta crocuta*	64	1.93%	0.64
Steenbok	*Raphicerus campestris*	35	1.05%	0.35
Tree squirrel	*Paraxerus cepapi*	1	0.03%	0.01
Vervet monkey	*Chlorocebus pygerythrus*	9	0.27%	0.09
Warthog	*Phacochoerus africanus*	23	0.69%	0.23
Waterbuck	*Kobus ellipsiprymnus*	0	0.00%	0

APPENDIX IV: Cheetah Identity kit for all identified individuals within and adjacent to NOTUGRE

Each cheetah was assigned a unique identity code consisting of two letters and a number. The first letter referred to the species (C = Cheetah), the second letter, the sex (F = Female; M = Male; US = Unknown sex). The number referred to the position in the identification sequence, but is separate for males and females. CF1 and CF5 were initially believed to be different cheetahs but were then confirmed to be the same individual and reclassified as one cheetah CF1, thus CF5 was removed from the sequence to avoid confusion. The individual profiles are shown below for each cheetah using the best available photographs in which the spot patterns are clearly visible. Both left hand side and right hand sides photographs of the cheetahs are included if available. Photos provided by volunteers of the cheetah photographic survey.

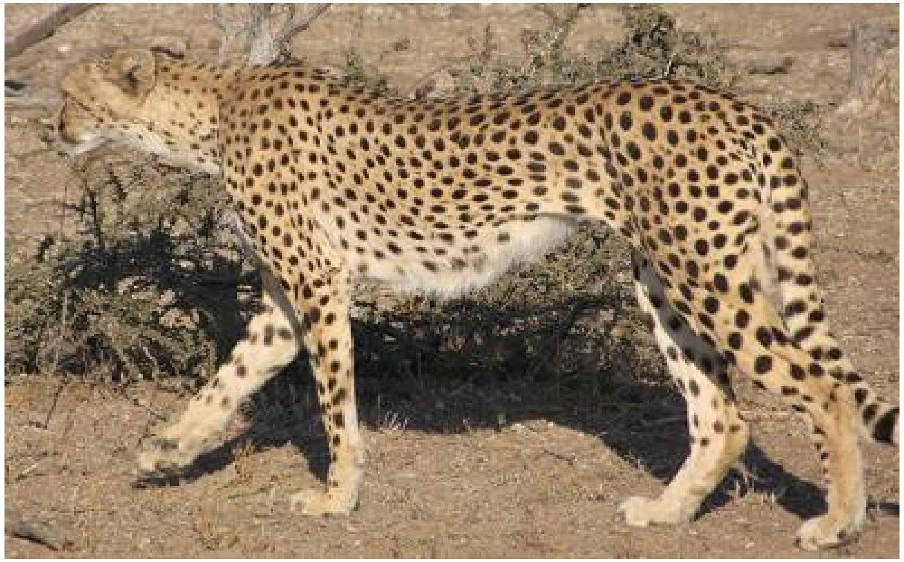
FEMALE CHEETAHS

RIGHT SIDE

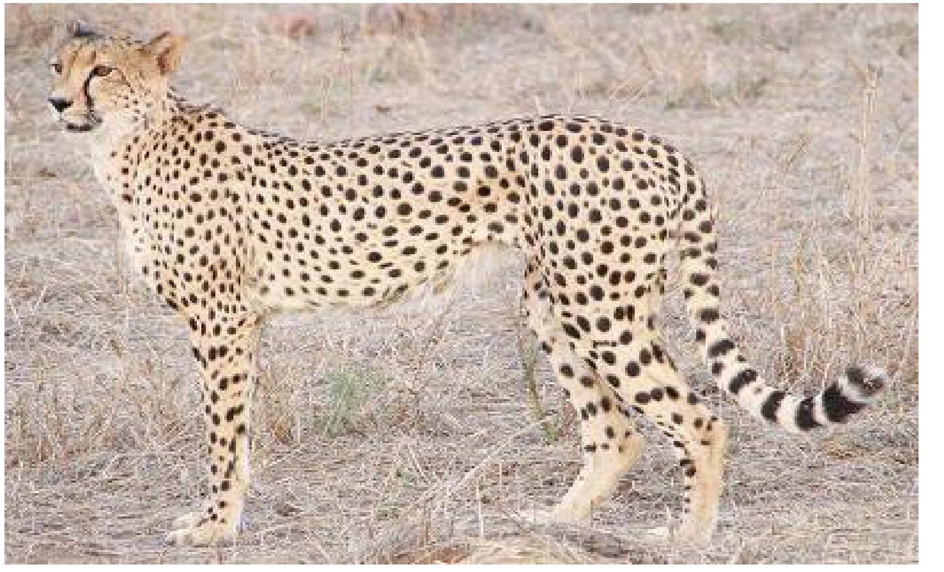
CF1

LEFT SIDE

CF2



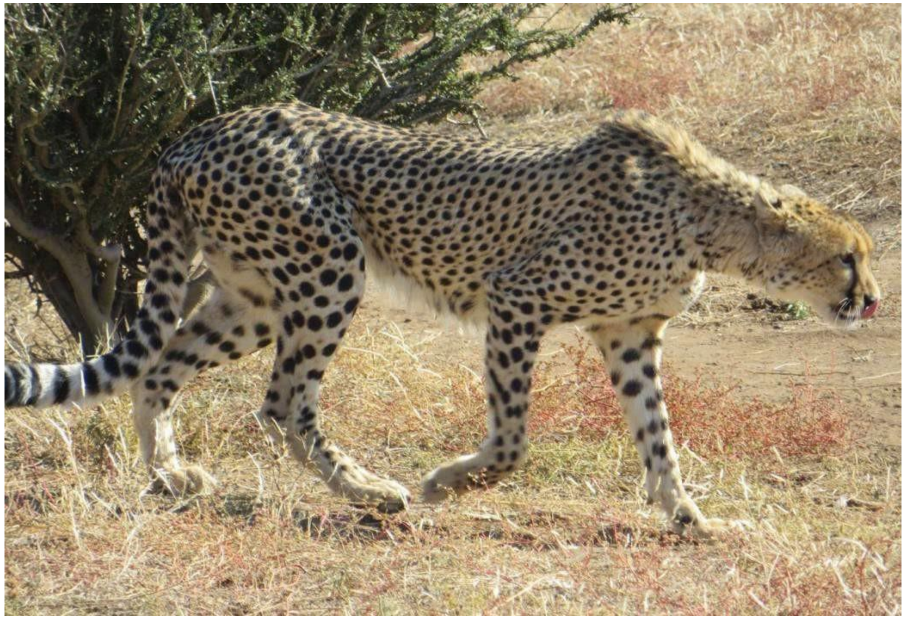
RIGHT SIDE **LEFT SIDE**



CF3 (Mapula)

RIGHT SIDE **LEFT SIDE**

CF4



CF6

RIGHT SIDE **LEFT SIDE**

CF7

CF8

RIGHT SIDE **LEFT SIDE**

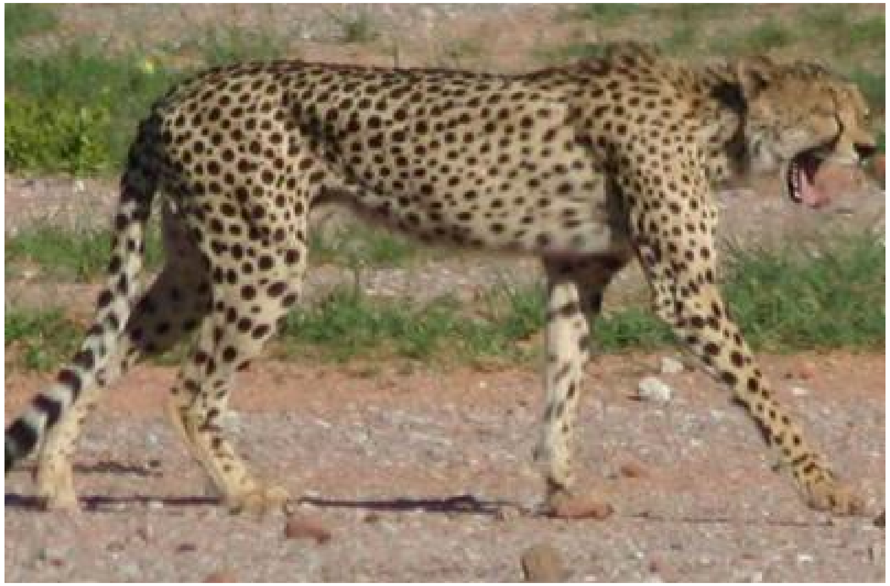
CF9

CF10

RIGHT SIDE | LEFT SIDE

CF11

CF12

RIGHT SIDE	LEFT SIDE
CF13	
	No image available
CF14	
No image available	

RIGHT SIDE	LEFT SIDE

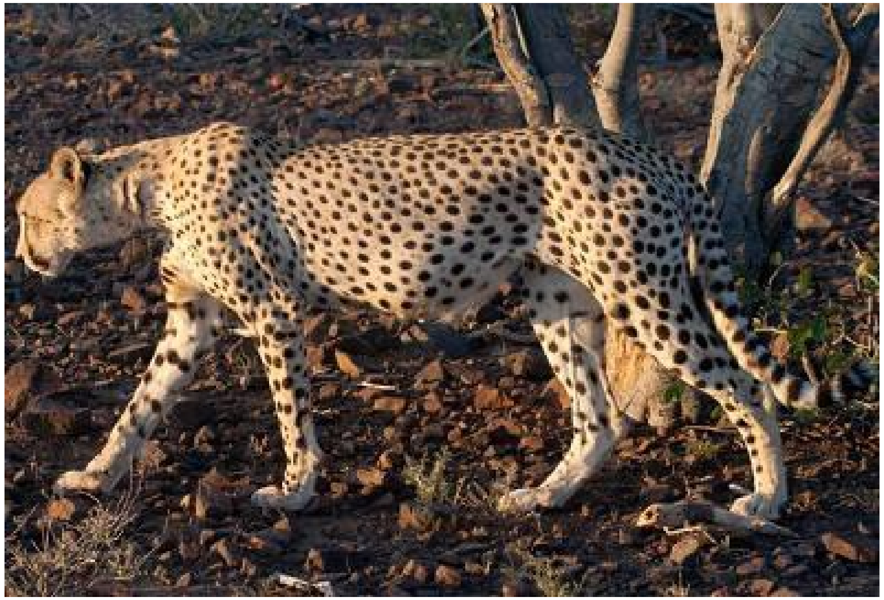
CF15

MALE CHEETAHS

CM1	
RIGHT SIDE	**LEFT SIDE**

CM1

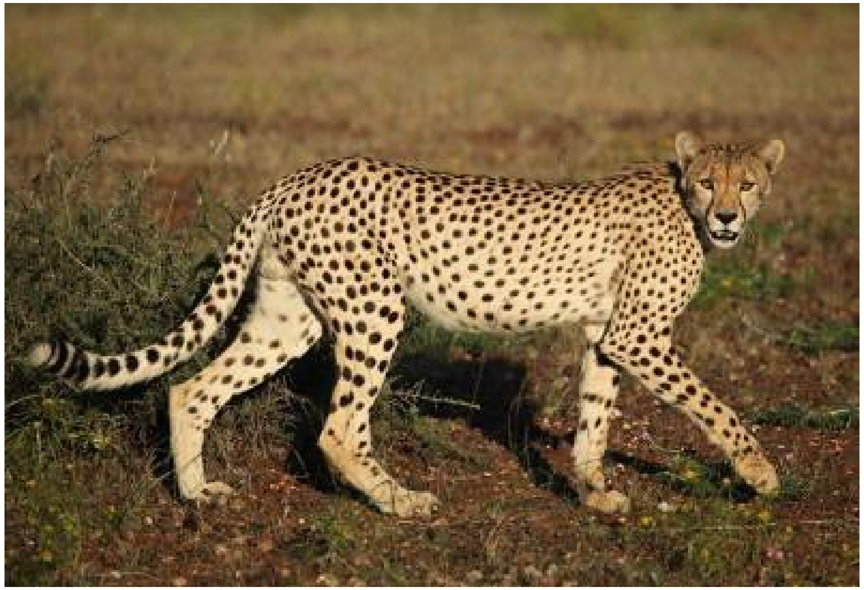
CM2



RIGHT SIDE	LEFT SIDE

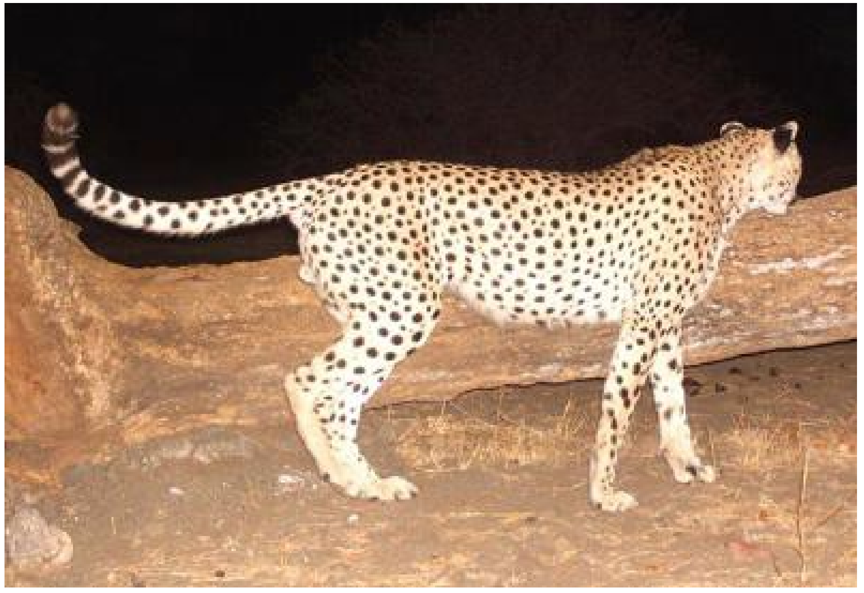
CM3

CM4



RIGHT SIDE **LEFT SIDE**

CM5



CM6

RIGHT SIDE

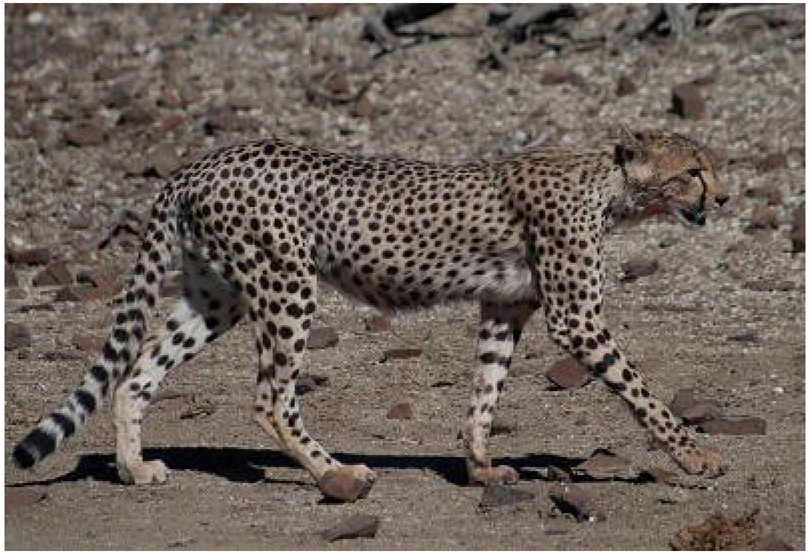
LEFT SIDE

CM7

CM8

RIGHT SIDE **LEFT SIDE**

CM9



CM10

RIGHT SIDE **LEFT SIDE**

CM11

CM12

RIGHT SIDE **LEFT SIDE**

CM13

No image available

CM14

RIGHT SIDE	LEFT SIDE

CM15

No image available

CM16

No image available

RIGHT SIDE

LEFT SIDE

CM17

No image available

CM18

No image available

UNKNOWN SEX

CUS1

CUS2

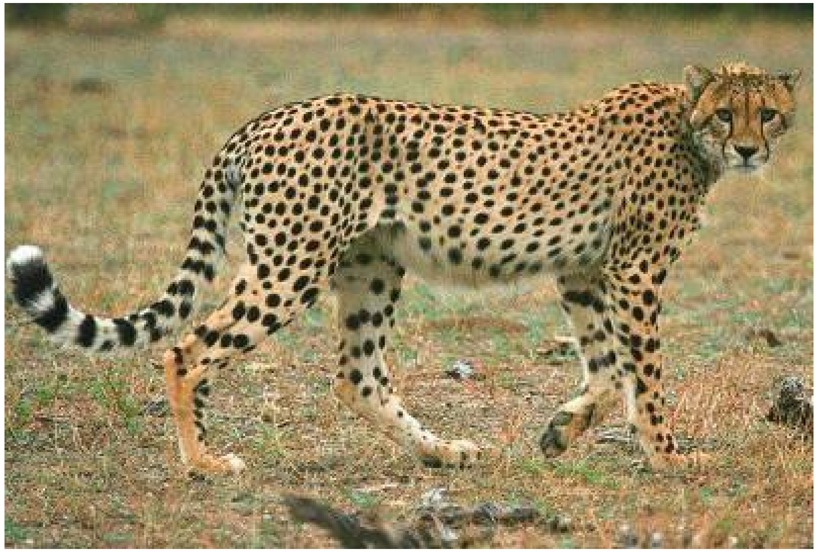
CUS3

CUS4

CUS5

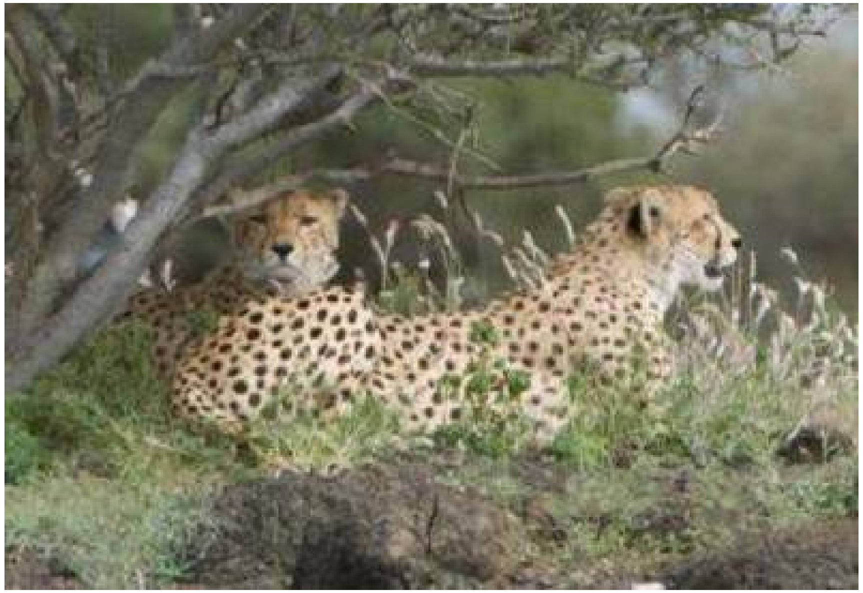
CUS6

CUS7



CUS8

CUS9



CUS10

CUS11

CUS12

CUS13

Appendix V: Description of all individual cheetahs identified during the photographic census; data period: January 2006-November 2013. The identification number, group name, number of sightings, availability of left hand side (LHS) and right hand side (RHS) profiles are shown and estimated date of birth (year and month), approximate age and relatedness is shown for individuals seen continuously over > 1 year. Age classes (A = Adult; C = Dependent cub/sub-adult) are given for individuals at first sighting. The age of cheetahs are calculated for the end of 2013.

Cheetah ID	Group name	No of sightings	LHS	RHS	Age class	YOB	Month	Age	Family relation	First sighted	Last sighted	Country sighted
CF1		26	Y	Y	A	Prior to 2005		>9	Mother of CM1, CM2; CUS10 (first litter); CM4 (second litter)	17/01/2006	08/01/2013	BOTS
CF2	Family of 2	28	Y	Y	A	Prior to 2006		>8	Mother of CF10	18/11/2008	29/06/2012	BOTS
CF3	Family of 6	155	Y	Y	A	2006	Jan-Feb	7.5	Mother of CF4; CF9; CF11; CM9; CM11 (first litter); CF8 (second litter)	01/10/2008	03/11/2013	BOTS
CF4	Family of 6	87	Y	Y	C	2011	Apr-May	2.5	Daughter of CF3; sister of CM9, CM11, CF9,CF11	26/10/2011	22/10/2013	BOTS
CF6	Female with scar	7	Y	Y	A	Prior to 2010		>4	Mother of CF7	09/02/2012	12/01/2013	BOTS
CF7	Female with scar	6	Y	Y	C	2011	Jul -Aug	2	Daughter of CF6	09/02/2012	12/01/2013	BOTS
CF8		58	Y	Y	C	2013	May	7 m	Daughter of CF3	24/05/2013	03/11/2013	BOTS
CF9	Family of 6	79	Y	Y	C	2011	Apr-May	2.5	Daughter of CF3; sister of CM9, CM11, CF4,CF11	26/10/2011	12/08/2013	BOTS
CF10	Family of 2	16	Y	Y	C	2010	Jun-July	3.5	Daughter of CF2	12/2/2011	05/07/2012	BOTS
CF11	Family of 6	60	Y	Y	C	2011	Apr-May	2.5	Daughter of CF3; sister of CM9, CM11, CF4,CF9	26/10/2011	07/12/2012	BOTS
CF12		3	Y	Y	A	Prior to 2005			Mother of CM14 & CM15	06/08/2006	31/01/2007	BOTS
CF13		1	N	Y	A	N/A				06/01/2007	06/01/2007	RSA
CF14	Moyo family	1	Y	N	A	N/A				20/06/2013	20/06/2013	RSA
CF15	Moyo family	1	Y	Y	SubA	2012			Daughter of CF14; sister of CM18	20/06/2013	20/06/2013	RSA
CM1	Coalition of 3	78	Y	Y	Cub	2006	Jun-Jul	7.5	Son of CF1; brother of CM2	13/12/2006	28/10/2013	BOTS/ZIM
CM2	Coalition of 3	79	Y	Y	Cub	2006	Jun-Jul	7.5	Son of CF1; brother of CM1	13/12/2006	28/10/2013	BOTS/ZIM
CM3	Coalition of 3	75	Y	Y	A	Prior to 2006		>7		01/05/2008	28/10/2013	BOTS/ZIM
CM4	5		Y	Y	SubA	2009	Jun -Aug	4.5	Son of CF1	11/08/2010	15/05/2011	BOTS
CM5	Coalition of 2	5	Y	Y	A	Adult			Coalition with CM6	28/06/2013	29/10/2013	BOTS
CM6	Coalition of 2	6	Y	Y	A	Adult			Coalition with CM5	28/06/2013	29/10/2013	BOTS

CM7	Venetia coalition	7	Y	Y	A	Adult			Coalition with CM8	04/05/2013	09/09/2013	RSA
CM8	Venetia coalition	1	Y	Y	A	Adult			Coalition with CM7	16/08/2013	16/08/2013	RSA
CM9	Family of 6	74	Y	Y	Cub	2011	Apr-May	2.5	Son of CF3; brother of CM11, CF4, CF9, CF11	26/10/2011	14/04/2013	BOTS
CM10		7	Y	Y	A	2006	Jan-Feb	2.5	possibly brother of CF3	01/10/2009	12/11/2009	BOTS
CM11	Family of 6	63	Y	Y	Cub	2011	Apr-May		Died Son of CF3; brother of CM9, CF4, CF9, CF11	26/10/2011	16/12/2012	BOTS
CM12	Vehmbe cheetah	1	Y	Y	A	Adult				15/09/2012	15/09/2012	RSA
CM13		1	Y	N	A	Adult				16/12/2006	16/12/2006	BOTS
CM14		3	Y	Y	SubA	2006	Mar -May		Son of CF12; brother of CM15	25/01/2007	31/01/2007	BOTS
CM15		2	Y	N	SubA	2006	Mar -May		Son of CF12; brother of CM14	25/01/2007	31/01/2007	BOTS
CM16		2	N	Y	SubA	2006 - 2007			Brother of CM17; CUS6; CUS7	02/05/2008	29/06/2008	BOTS
CM17		1	N	Y	SubA	2006 - 2007			Died Brother of CM16; CUS6; CUS7	02/05/2008	02/05/2008	BOTS
CM18	Moyo family	1	Y	N	SubA	2012			Son of CF14; brother of CF15	20/06/2013	20/06/2013	RSA
CUS1		2	Y	N	A	Prior to 2007				22/03/2008	18/06/2008	BOTS
CUS2		1	Y	N	A	Prior to 2006				18/03/2006	18/03/2006	BOTS
CUS3		1	N	Y	A	Prior to 2005				17/01/2006	17/01/2006	BOTS
CUS4		1	N	Y	SubA	Prior to 2005				17/01/2006	17/01/2006	BOTS
CUS5		1	Y	N	SubA	Prior to 2005				17/01/2006	17/01/2006	BOTS
CUS6		2	Y	N	SubA	Prior to 2008			Died	02/05/2008	02/05/2008	BOTS
CUS7		1	N	Y	SubA	Prior to 2008			Died	02/05/2008	02/05/2008	BOTS
CUS8		1	N	Y	A	Prior to 2002				12/01/2003	12/01/2003	BOTS
CUS9		1	N	Y	A					24/03/2006	24/03/2006	BOTS
CUS10		1	N	Y	Cub	2006	Jun - Jul		Died sibling of CM1, CM2	13/12/2006	13/12/2006	BOTS
CUS11		1	Y	N	A				sibling of CUS12	21/08/2007	21/08/2007	BOTS
CUS12		1	Y	N	A				sibling of CUS11	21/08/2007	21/08/2007	BOTS
CUS13		1	Y	N	A					01/03/2012	01/03/2012	BOTS

APPENDIX VI: Questionnaire

Date	**Interviewer**		**Q.No:**
Coordinates	**S:**	**E:**	**Region/Village:**

Section A: General Details

1.	Name			Anonymous		
2.	Are you:	owner	herdman	worker	other	
3.	If you are the owner: do you own/lease/rent the land?					
4.	How long have you been in the area?					

Section B: Socio-Economic Details

5.	Year of birth? (age)	6.	School level?		
7.	What is the main source of family income?		Smallstoc	Cattle	Hunting
			Employm	Crops	Tourism
			Other………………………		
8.	What role do livestock play, if it is not the main source of income?				

Section C: Farm Details

9.	How many persons live on cattlepost?						
10.	Vegetation type:		11.	Can you give a percentage for each?			
	Open grassland and pans		Sparse bush 10-35%	Medium bush 35-65%		Thick bush 65-100%	don't know
12.	Have there been changes in the habitat over time?					yes	no
13.	Specify (how / over what time period)						don't know
14.	How many animals do you keep?						
		Number	Breeds			Number	breed
	Cattle			Dogs			
	Sheep			Other			
	Goats						
	Horses						
	Donkeys						
	Chicken						
15.	Have you had major increase / decrease in livestock? How, why and over what time period?						
16.	What are the main problems encountered by livestock farmers? *Rank importance: max 3*						
	diseases		insufficient grazing			theft	
	drought		poor quality grazing			snares	
	infertility		low yields			vet fence	
	losses due to predators		unreliable market			miscarriage	

Section D: Farm Management

17.	Please explain how your **cattle** are tended to? (e.g. people as herders, kraal, dogs) Night: Day:		
18.	Please explain how your **goats & sheep** are tended to? Night: Day:		
19.	**If kraal** Time out of kraal in morning:	Time in Kraal in afternoon:	
	Cattle: Goats & sheep:	Cattle: Goats & Sheep :	

20.	What is your kraal design? (stones, wooden posts, acacia bush, fencing) Cattle:		
21.	Distance of kraal from homestead:		
22.	Height:	Gaps in kraal:	Stems: in/out
23.	Goats and sheep:		
24.	Distance of kraal from homestead:		
25.	Height:	Gaps in kraal:	Stems in/out
	Calving/lambing		
	During calving / lambing, do you:		
26.	Bring calving animals closer to homestead?	yes	no
27.	Check on livestock more often than before?	yes	no
28.	Keep careful records?	yes	no
29.	Kraal all livestock at night?	yes	no
30.	Kraal young calves / kids during the day?	yes	no
31.	Use a maternity / calving kraal?	yes	no
	Other?		
32.	Do you have a herder with livestock? (Specify cattle/goats/sheep)	yes	no
33.	How many do you have per number of livestock?		
34.	Who are they (paid workers, children)?		
35.	What do they do?		
36.	Are they effective?		

37.	Distance travelled to furthest grazing:	1km	1-5km	5-10km	>10km
38.	Do you have a dog with livestock? (Specify cattle/goats/sheep)			yes	no
39.	What breed?				
40.	Size of dog:	Small (cat size)	Medium	Large (Anatolian/German shepherd)	
41.	How many do you have per number of livestock?				
42.	Are they effective?				
43.	How does the dog protect livestock:	barking	chasing	killing predator	other

44.	Does the dog live with livestock day and night?

Section E: Wildlife Details

45.	Please detail which game species exist in the area:		
Species	Status (absent, rare, common, very common)	Trends over past 10 years (increase, decrease, stable)	
Kudu			don't know
Impala			don't know
eland			don't know
zebra			don't know
wildebeest			don't know
duiker			don't know
steenbok			don't know
warthog			don't know
hares			don't know
guineafowl			don't know
other			don't know
other			don't know
46.	Do you have explanations for any changes in the numbers?		don't know

Section F: Predator Details

47.	How often do you see this predator?				
Species	how often	visual, tracks, calls	increasing, decreasing, stable	KEY	
Lion				never	0
Cheetah				< once /year	1
Leopard				once /year	2
Spotted Hyena				a few times a year	3
Brown Hyena				every few months	4
Wild dog				once/month	5
Baboon				every week	6
Jackal				everyday	7
Honey Badger				don't know	-
Other					
48.	Do you have explanations for any changes in the numbers?			don't know	

Section G: Cheetah Details

49.	In the past 18 months, how many times did you see cheetahs (**S**)? Or have report of cheetah sightings? (**R**) Describe when, where, the numbers of adults and cubs and what was its activity.				
	Date	Group Composition (number, age, sex)	Location (habitat type)	Activity	Time of day
50.	What is the maximum number of cheetahs you have seen at the same time?				

	Adults		cubs (size)			don't know
51.	Where did you see them?					
	Are cheetahs on your farm:		permanent	seasonal		
52.	What time of the year?					
53.	If you think numbers are increasing/decreasing, what are the reasons in your opinion for such as change?					
						don't know
54.	Are cheetahs protected in	yes	don't know			
55.	Are cheetahs endangered	yes	don't know			
56.	Have cheetahs any value for you?	yes	no			don't know
57.	Specify (traditional, totem, medicine, food, beauty, etc. OR danger, threat, problem, etc.)					
58.	Have your neighbours seen cheetahs?			yes	no	don't know

Section H: Predation and conflicts

59.	Do you lose livestock to predators			yes	no	don't know
60.	Classify the predators, according to level of problem:			Rank: 1: biggest problem; 10: least problem		
	Spotted hyena		Lion		Honey	
	Cheetah		Wild dog		Eagle	
	Leopard		Jackal		Hawks	
	Brown hyena		Baboon		Other…	
61.	If you had a problem with predators in the last 18 months, describe:					
	Date or Season	Animals killed or injured (no, spp, age, sex)	Predators responsible (number, spp, age, sex)	How it was identified (visual (by who), spoor by kill, killing bites, feeding style)	Time of day of incident	Location (near water, in kraal, out)
62.	Are these incidents from accurate records or from memory?					
63.	What do you do to protect cattle from predators?					
64.	Kraal			yes	no	
65.	Use people as herders			yes	no	
66.	Guard animals (e.g. dogs)			yes	no	
67.	Chase away/make noise			yes	no	
68.	Use poison baits			yes	no	
69.	Other (explain)					
70.	What do you do to protect goats/sheep from predators?					
71.	Kraal			yes	no	
72.	Use people as herders			yes	no	
73.	Guard animals (e.g. dogs)			yes	no	
74.	Chase away/make noise			yes	no	
75.	Use poison baits			yes	no	
76.	Other (explain)					
77.	Are measures you take to protect livestock effective?			yes	no	
78.	If not, why?					
79.	What are the common circumstances of attacks?					
	Day		Inside kraal		Herder	
	Night		Outside kraal		No herder	

80.	Are losses to predators seasonal?	yes		no	don't know
81.	Which season?				
82.	Have you lost animals in the past 18 months due to other causes than predators? Specify: number/species				
	If no numbers: rank importance: max 3				

	Diseases		Starvation		Miscarriage	
	Calving		Theft		Other	
	Accidents		Crossing vet fence		Other	
83.	Can you give an approximate value for these losses in the last 18 months?					
	During your time in the area is the problem with predators:			increasing	decreasing	stable
84.	Can you give reasons why?					
85.	What do you do when you have a loss to a predator? (nothing, scare off predator, report to wildlife officer, kill predator, other)					
86.	Did you ever have to remove predator?			yes	no	
	Details: How? When? (live trap, shoot, poison)				don't know	
87.	Have you contacted Wildlife officer for assistance?		yes	no	don't know	
88.	Details?					

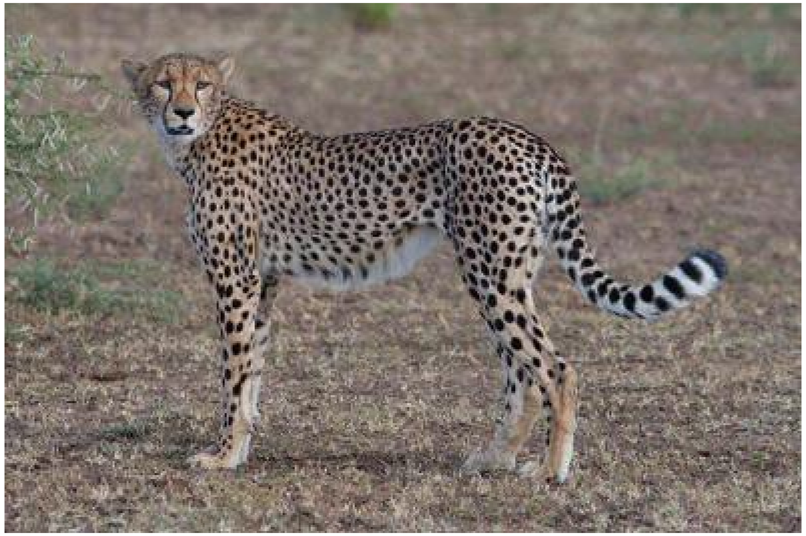
Section I: Attitudes

89.	What do you think about sharing the land with predators?					
	Benefit to farm	Like them	Dislike them	Kill when see	Other	don't know
90.	Why?					
91.	Do you think wildlife is a national resource to be protected?				yes	no
92.	Why?					
93.	Whose responsibility do you think predator/livestock conflict belongs to?					don't know
	Owner	Herders	Government	NGO's	Game reserves	Other
94.	Do you see any solutions for the survival of predators on farmlands?					
	Improve farm management	Decrease numbers			Translocate	Other
	Trophy hunting	Compensate			Tourism	

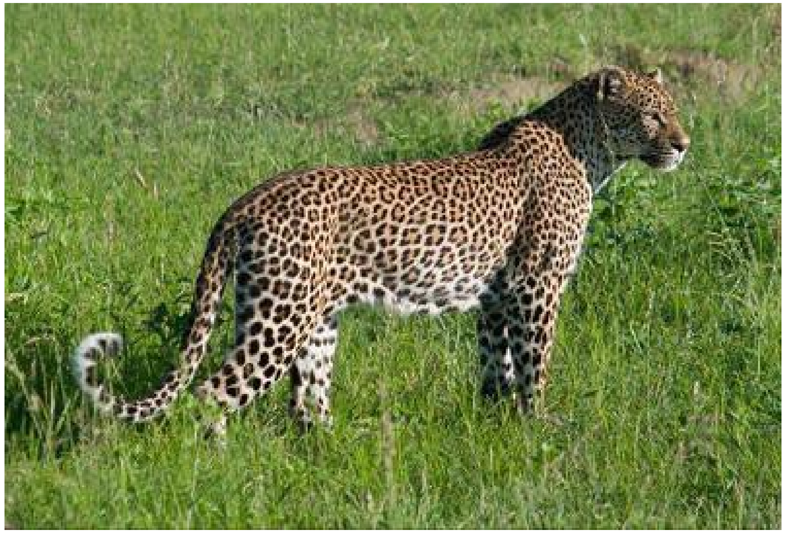
Contact address:					
Tel number:					
Ranking					
Precision		Co-operative attitude			Total
Consistency		No wrong or doubtful info			/4
Predator identification		Cheetah	Leopard	Lion	Wild dog
		Spotted hyena	Brown hyena	Domestic dog	Black-backed jackal

APPENDIX VII: Predator images used in the questionnaire survey as supplementary material to assess knowledge of the common predators.

Cheetah (*Acinonyx jubatus*) (Photo: Eleanor Brassine)



Leopard (*Panthera pardus*) (Photo: Eleanor Brassine)



Lion (*Panthera leo*) (Photo: Eleanor Brassine)

African wild dog (*Lycaon pictus*) (Photo: Eleanor Brassine)

Spotted hyena (*Crocuta crocuta*) (Photo: Eleanor Brassine)

Brown hyena (*Hyaena brunnea*) (Photo: Martin Harvey)

Domestic dog (*Canis lupus familiaris*) (Photo: Mathilde Brassine)

Black-backed jackal (*Canis mesomelas*) (Photo: Eleanor Brassine)

Appendix VIII: Examples of the different kraal designs and materials utilised by livestock farmers.

Horizontal poles (split-rail fence) (Photo: Eleanor Brassine)

Vertical wooden posts (Photo: Eleanor Brassine)

Acacia branches (Photo: AWF/Nakedli Maputla)

Combination of wooden posts and fencing (Photo: Eleanor Brassine)

Fencing (Photo: Eleanor Brassine)

Fencing with diamond mesh (Photo: Eleanor Brassine)